Chapter 4.
Stream Fish Indexes

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INTRODUCTION

The approach of measuring the biological integrity of rivers and streams using a quantitative index of biological integrity (IBI) based on fish assemblages[7] was first described by Karr (1981). Since then the IBI approach has been widely applied to warm water streams and rivers in North America and elsewhere in the world. However, the application of the IBI to the cold water rivers and streams of the western USA has been limited (Simon 1999). The purpose of this paper is to describe the adaptation and testing of the IBI approach to cold water streams in Idaho's forested and rangeland ecoregions. The approach draws heavily from similar work to develop an IBI for cold water rivers in Idaho (Mebane 2000).

Waters that have biological integrity, as used here, are those that have "… a species composition, diversity, and functional organization comparable to that of natural habitats of the region" (Frey 1977). In the development of quantitative indices, these components of biological integrity (composition, diversity, and functional organization) are measured, and are intended to relate fish assemblages to other biotic and abiotic components of the ecosystem (Karr et al. 1986, Simon 1999). Comparisons to natural habitats of the region require classifying streams into relatively homogenous categories of fish assemblages. The natural fish assemblages are assumed to be shaped by broadscale factors such as selective extinctions during the late Pleistocene; recolonization patterns; long-term zoogeographic barriers such as waterfalls; broad climatic conditions (e.g. Benke 1992, Moyle 1994); intermediate, or stream scale climatic and geomorphological factors such as stream gradients (e.g. Kruse et al. 1996); and site scale features such as adequate resting refugia from canopy cover or pools in desert or forest trout streams (Li et la 1994, Herger et al. 1996). Only at the finest scale (stream or site scale), are human-caused disturbances assumed to affect the natural fish assemblage. The intermediate and broadscale factors are assumed to be important for classifying stream habitats in order to be able to detect human-caused effects to habitat or water quality.

Karr et al. (1986) developed the IBI for warm water Midwestern streams (i.e. waters too warm to support salmonids). Their original IBI consisted of 12 metrics that reflected fundamental ecological components of stream fish assemblages, which included measures of: taxonomic richness, habitat and trophic guild composition, and individual health and composition. Following this lead, fish IBIs have been developed and published for many

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[7] The terms "assemblage" and "community" have been used interchangeably in bioassessment literature (e.g. Matthews and Hein 1987, Simon 1999a). Here, the term "assemblage" is used to mean a subset of a biological community composed of a single phylogenetic group, e.g. fish, arthropods (i.e. benthic macroinvertebrates), or diatoms.

regions of temperate North America and in other parts of the world (Simon 1999). Because of the substantial differences in fish faunas in areas outside the midwestern United States where the original IBI was developed, other workers have substituted regional metrics for the original 12 metrics. Most have maintained the ecological structure of the original IBI. Due to all the iterations, the term "IBI" should be thought of as a family of related indices, rather than a single index (Simon and Lyons 1995).

Most IBI versions have been developed for warm water streams rather than for cold water systems. Three significant factors complicate IBI development in cold water systems. First, the cold water streams of western North America have much lower species richness than do the cool or warm water systems. Selective extinctions in the late Pleistocene are still a major factor shaping modern, western fish assemblages. In contrast, the vast Mississippi drainage served as a refuge for fishes, avoiding extinctions, and possibly increasing speciation. The least disturbed Western streams have a nearly universal cold water-adapted fish assemblage: salmonids (*Oncorhynchus/Salvelinus* sp.), sculpin (*Cottus* sp.), sucker (*Catostomus* sp.), and dace (*Rhinichthys* sp.). In contrast, streams in the Mississippi drainage may have 40 to 50 native species (Moyle 1994). A second complicating factor is that species richness in warm water streams declines as habitats or water quality are degraded. In contrast, as cold water systems are degraded, species richness often increases as native cold water fishes decline, making their habitats vulnerable to invasion by facultative cool water native species or non-indigenous species, most of which were introduced from the Mississippi drainage (Moyle 1994). A third complicating factor is the homogenization of fish assemblages by extensive stocking of salmonids, which further blurs the already depauperate cold water western fish assemblages (Maret et al. 1997;. Rahel 2000).

These factors have slowed the development of a generally applicable cold water IBI for western streams and rivers. Despite these limitations, river fish assemblages are socially and ecologically important and should be evaluated in their own right. Regional state water quality standards and the national Clean Water Act explicitly call for protection of fish and fisheries. Direct assessment of the fish assemblage is more relevant to its management than surrogate approaches based on other assemblages or chemical and physical criteria alone.

METHODS

Since this report is longer than most research papers and presents diverse results, some methodological details are split up and reported with the respective results.

The analysis included descriptive and statistical approaches to classifying generally like and unlike stream types, and evaluating candidate metrics (variables that respond to water quality or habitat degradation). The latter step included examining patterns of potential response variables across undisturbed or minimally disturbed stream types, and across a range of sites with apparent human disturbance. The response variables (metrics) were converted to unitless scores that were then added together to form composite indexes. The indexes were then tested to see if they could discriminate between reference and impaired sites. The index results were additionally compared to amounts of fine sediment in stream channels and a habitat index. Excessive amounts of fine sediment and habitat degradation are common causes of stream degradation. Thus, there should be some relationship between index scores and sediment and habitat measurements.

Data

Data were compiled primarily from the following sources:

- Idaho Department of Environmental Quality (DEQ) Beneficial Use Reconnaissance Program (BURP)
- Idaho Department of Fish and Game (IDFG) Resident Species Research Program
- Idaho State University, Stream Ecology Center (Robinson and Minshall, 1995)
- U.S. Geological Survey National Water Quality Assessment (NAWQA) Upper Snake River, Northern Rockies, Columbia Plateau, and Salt Lake study units[8] (limited to sites located on small streams in the ecoregions of interest)

Criteria for use of data were that they include at least single pass electrofishing data, with all fish species captured, identified, and recorded. This requirement excluded many surveys that recorded salmonids only. Other needed information included a physical description of the site (location, size, elevation), lengths of salmonids and sculpins, and electrofishing effort (duration in seconds).

Classification of Stream Types

Like taxonomy of organisms, classification attempts to distinguish and group distinct environments, communities, or ecosystem types. The objective of stream type classification is to group places where living systems are similar at higher taxonomic and ecological levels in the absence of human disturbance, and where the biological responses are similar after human disturbance (Karr 1999). Classifications are based on natural history, biogeography, and data analysis. In particular, a stream classification based on river morphological characteristics is helpful in predicting stream behavior, extrapolating data from one stream to another of similar character, and providing a consistent and reproducible frame of reference (Maret et al. 1997).

Major factors in grouping stream types

Ecoregion – Stream types were initially classified into two broad groups, forest and rangeland streams. Burton et al (1991) concluded that on a regional scale in Idaho, there are two major streams types that are clearly differentiated on factors limiting fish populations: 1) Stream and riparian systems dominated by forest overstory, and 2) those dominated by grass and shrub riparian vegetation. Forest canopy dominated streams occur primarily in mountain settings in Idaho and occur generally on gradients of more than 1.5 percent. Grass

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[8] http://montana.usgs.gov/nrok/nrokpage.htm; and http://wwwidaho.wr.usgs.gov.nawqa/usnk_home.html

and shrub streams occur in the intermontane valleys, mountain meadows, and plains, and generally have gradients of less than 1.5 percent.

In forested mountain streams, salmonid populations are limited primarily by habitat structure. Physical habitat diversity is apparently the key to population size because in steeper gradient streams, resting areas and refugia are physically limited (Bjornn and Reiser 1991, Herger et al. 1996). Canopy closure in forested mountain streams may often limit primary production and the availability of drifting prey to fish due to the lack of light energy penetration in these streams. In contrast, most rangeland streams are located in meadows and valleys mostly at lower elevations in Idaho. Stream gradients are predominantly less than two percent, and natural riparian vegetation is dominated by grasses and shrubs, with little or no woodland overstory. In rangeland streams, light energy inputs and resting and feeding are not generally limiting (Burton et al. 1991). Beyond these differences in natural stream ecosystems, forest and rangeland streams may have different responses to environmental degradation. In forested stream systems, removal of the stream canopy by logging may increase local primary production, macroinvertebrate abundance and diversity, and abundances of salmonids, sculpin, and amphibians (Hawkins et al. 1983, Murphy and Meehan, 1991, Meehan 1996). In contrast, in the high desert trout streams of the inland Northwest, trout biomass was negatively correlated with loss of canopy due to grazing, despite increases in invertebrate productivity (Li et al. 1994).

Ecoregions of the Pacific Northwest correspond well with this forest/rangeland stream distinction. These ecoregions are based on perceived patterns of integrative factors including land use, landform, potential natural vegetation, and soils (Omernik and Gallant 1986). The seven ecoregions Omernik and Gallant mapped in Idaho can be grouped into predominately montane-forested ecoregions (Northern Rockies, Middle Rockies, Blue Mountains, and Wasatch-Uinta Mountains) and predominately desert basin-rangeland ecoregions (Snake River Basin/High Desert, Northern Basin and Range, Columbia Basin and Wyoming Basin). Henceforth, the terms “mountain,” “forest,” “basin,” or “rangeland” ecoregions are used to mean these groups. Fish assemblages in the forested streams are usually dominated by or exclusively composed of stenothermic trout and sculpin species (Burton et al. 1991, Maret el. 1997). Rangeland assemblages of the arid ecoregions of the Northwest are usually reported to include a mix of mesothermic minnows and suckers, and the stenothermal trout and sculpin species (e.g. Whittier et al. 1988, Hillman 1991, Pearsons et al. 1992, Tait et al. 1994, Maret el. 1997).

Elevation – I used elevation as a secondary factor to group the streams. Maret et al (1997) found that patterns of fish species distributions and assemblages did not correspond well with the mapped ecoregion boundaries. They suggested the reason may have been because many of their least-disturbed sample sites in rangeland (Snake River Basin and Northern Basin and Range) ecoregions were located near transitional areas between ecoregions and may have expressed characteristics of adjacent ecoregions. Robinson and Minshall (1995) concluded that *within* both the mapped Snake River Basin and Northern Basin and Range ecoregions, reference streams could be distinguished into upland and lowland streams by gradient, elevation, and whether the watersheds were typically forested or sagebrush/grass. The upland streams shared features of the Northern Rockies streams, in that they had higher gradients and more forest cover than the lowland streams, which were more typical of the rangeland ecoregions. Most of the stream sites they considered lowland sites were below about 1600m and 1750m for the Snake River Basin and Northern Basin and Range ecoregions respectively. For this study, streams sites that were located above these elevations in these two ecoregions were considered to be "forest" streams. An elevational threshold of about 1200m was selected for this distinction for the Columbia Basin ecoregion of northern Idaho. I am not aware of any ecoregional comparative studies similar to Robinson and Minshall (1995) for this area. Instead, I selected 1200m based upon review of vegetation patterns shown on topographical maps, discussion with regional biologists, and the general ecological rule of thumb that in temperate latitudes, 1° of latitude is approximately equal 200m of elevation change (e.g. Flebbe 1996)[9] Latitude differences between Columbia Basin and southern Idaho rangeland ecoregion sites range from about 1 to 5°. Ecoregions and these ecoregion combinations are illustrated in Figure 4-1.

Stream size – Stream order or stream size is a major determinant for species richness, abundance, and life history use by fish (Platts 1979, Li et al. 1987). In theory, stream size should be an important factor in determining the fish assemblage in both rangeland and forested streams. However, few perennial rangeland streams could be considered in a reference condition over a significant longitudinal gradient. This prevented evaluation of different stream sizes as a classification feature. Other than the original distinction that this study is focused on wadeable perennial streams (i.e. no large rivers), stream size and fish assemblage relationships were not investigated in rangeland streams. In the forest streams, stream size and elevation are considered in classifications and evaluation of candidate metrics.

[9] Before finding Flebbe's reference to this rule of thumb of 200m per degree of latitude, I estimated latitudinal-elevational changes in Idaho to be about 170m per degree of latitude, which suggests this is a pretty good rule of thumb. I made this estimate by comparing alpine timberline elevations on topographic maps across 7° of latitude ranging from the Selkirk Mountains (≈1830m at 49°N) to the Albion Range in the Northern Basin and Range ecoregion (≈2990m at 42°N), including several mountain ranges in between (the Bitteroots, Boulder Mountains, Salmon River Mountains, and the Tetons).

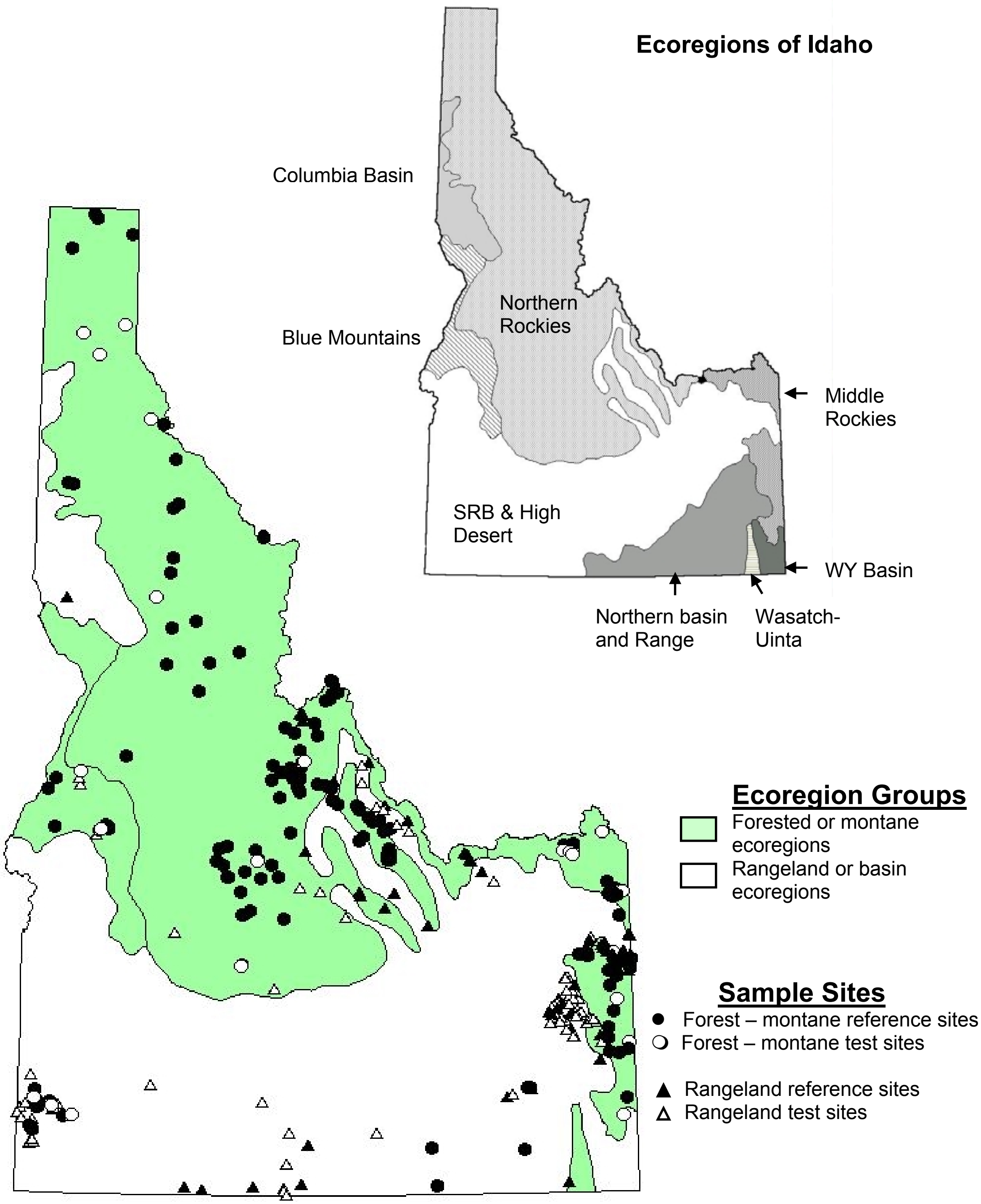


Figure 4-1. Ecoregions of Idaho (inset-top), combined ecoregion groups (bottom), with fish assemblage sample sites. Some sites not shown, due to missing coordinates.

Evaluation and Selection of Candidate Metrics

Selecting and testing candidate metrics involved three major steps:

- Compiling sampling data from a variety of sites with different types and intensities of human disturbance.
- Calculating attributes (metrics) that, according to literature reports, would be likely to show a relationship to human disturbances.
- Searching for empirical relationships, i.e. quantitative change across a range of conditions.

Candidate metrics

Many measures of composition, diversity, and functional organization of fish assemblages have been used or postulated for use to develop an IBI. Barbour et al. (1999) list 69 measures, and there are others beyond those. To focus the testing, I reviewed other efforts to modify the IBI approach to cold water or other species depauperate systems. Metrics used or proposed for indexes of biological integrity in several other cold water North American areas are listed in Table 4-1. Attempts to overcome the limited species richness include several common themes:

- Expanding the fish assemblage to a vertebrate assemblage by including amphibians
- Interpreting the ubiquitous salmonid populations through age class strength and abundance or biomass
- Interpreting sculpin populations, which are often sympatric with salmonids, but are not stocked or harvested
- Interpreting the composition of stenothermic and mesothermic fishes (i.e. obligate cold water and cool water tolerant fishes).

While these themes from previous investigations give insight into potentially useful metrics, in some cases the ecological interpretation of similar metrics have been contradictory or unexplained. For example, Fisher (1989) found significant correlations between increasing salmonid density, abundance and richness of amphibian species, and increasing timber harvest in their watersheds. Based on these correlations, he structured his index so that high species richness, abundance of amphibians, and high salmonid densities would indicate low biological integrity. In the Idaho qualitative index, dominance by benthic insectivores was assumed to be an indication of impairment (Table 4-1). However, a large portion of the natural fish assemblage in mountain streams can be considered benthic insectivores, since most juvenile salmonids occupy positions near the benthos and feed predominantly on benthic insects (Meehan and Murphy 1991), and the diet of all resident sculpin species are largely benthic insects (Bond 1963, Hendricks 1997, Zaroban et al. 1999). These examples caution us that metrics used in a multimetric index development should have both a theoretical basis and empirically tested performance. The ecological concepts and reasoning of each candidate metric are described in Table 4-2.

Table 4-1. Metrics from original IBI and suggested for cold water streams.
The symbols (-) or (+) indicate a metric for which decreased or increased value (respectively) was supposed to indicate degraded water quality.

Original IBI (Karr et al. 1986)	**Cold water rivers in Idaho (Mebane 2000)**	**Northwest Coast Range cold water streams (Howlin et al. ms.)**
# fish species (-) # sucker species (-) # darter species (-) # sunfish species (-) # intolerant species (-) % green sunfish (+) % omnivores (+) % insectivorous cyprinids (-) % top carnivores (-) % hybrids (+) abundance or catch per effort (CPUE) (-) % with anomalies (+)	# cold water native species (-) # sculpin age classes (-) % sensitive native individuals (-) % cold water individuals (-) % tolerant individuals (+) # non-indigenous species (+) % carp (+) # age classes for selected salmonid species (-) # Cold water individuals/ electrofishing effort (-) % with deformities, eroded fins, lesions, or tumors (DELT anomalies) (+)	# native vertebrate species (-) % introduced species (+) % anadromous (-) % cool water (+) % tolerant individuals (+) # native cold water individuals (-) % native cold water individuals (-) # age classes for selected salmonid and sculpin species (-)
Idaho streams (Chandler et al. 1993)	**Cold water Vermont streams (Halliwell et al. 1999)**	**Northern and central Idaho erosive (e) or depositional (d) streams (Fisher 1989)**
# intolerant species (-) # non-indigenous species (+) # salmonid species (-) # benthic insectivore species (-) % insectivores (-) % total anomalies (+) % salmonids (-) Total fish biomass (-) Salmonid biomass (-) % young-of-year salmonids (-) Jaccard community similarity coefficient (-)	Resident lotic species Benthic insectivorous species Sensitive resident (lotic) species % omnivore and generalist feeders % individuals as benthic + water column insectivores % individuals as top carnivores + trout # individuals as lotic residents % individuals with disease or anomalies	# fish species (e -, d +) # salmonid species (e -) # non-salmonid species (e -, d +) # introduced species (e -) % salmonid individuals (e +) Average salmonid length (e -) Average salmonid weight (e -) Salmonid density (+, d -) Salmonid biomass (d -) Fish biomass (d -) # amphibian species (e +, d -) Amphibian biomass (d +) # intolerant species (d -) % hybrid species (d +) Macroinvertebrate density (d +)
Idaho qualitative IBI (IDEQ 1996)	**Cold water Wisconsin streams (Lyons et al. 1996)**	**Snake River Basin and Northern Basin and Range ecoregions (Robinson and Minshall 1992)**
Intolerant individuals (-) Tolerant species dominance (+) Benthic insectivores dominant (+) Hybridization with introduced species (+) Salmonid age class structure (shift or fewer classes) (-) Prevalence of anomalies (+)	# intolerant species (-) % cold water species (-) % tolerant species (+) % salmonids that are brook trout (i.e. natives) (-) % top carnivores (-)	# salmonid species (-) # tolerant species (+) % tolerant individuals (+) % salmonidae (-) Salmonid biomass (-) Salmonid condition factor (-)

Table 4-2. Ecological concepts and reasoning of selected candidate metrics.

Assemblage richness and composition metrics
Number of cold water native species *or* Percent cold water native individuals Species richness frequently changes in response to environmental stress. This metric is limited to native cold water species to exclude confounding introduced or tolerant native species. Reference cold water streams typically have one to three native cold water species. As habitats shift from cold to cool water, total species richness may increase as cool and warm water species expand their range. Most fish assemblages appear to be more unstable and fluctuate more in terms of species abundances than in terms of species presence or absence (Rahel 1990).
Number of native species In perennial streams in the rangeland ecoregions (Snake River Basin, Northern Basin and Range, and Columbia Basin), richness in reference streams may be four to six (or more) native species. As with other temperate areas for which IBIs have been developed, this metric assumes stressed streams will lose native species.
Percent cold water individuals This metric acknowledges widespread establishment of non-indigenous trout populations that have become part of the resident fishery in Idaho. Introduced trout often displace native trout but are still intolerant of degraded water quality conditions. Low representation of cold water species may indicate degraded conditions.
Jaccard community similarity coefficient (JC) Departures of faunal composition from that expected to occur at a site is an intuitive, interpretable, and ecologically meaningful way of describing biological impairment; taxa composition must change before structural or functional attributes of an ecosystem change (Hawkins 2000). Metric assumes taxa composition is relatively stable at reference sites, measures degree of similarity in taxa composition between mean of reference sites and a test site. Cool water rangeland streams have higher taxa richness than cold water streams, suggesting measures of community persistence, or comparison of observed taxa presence to expected taxa presence may be useful. Jaccard coefficients values increase as the degree of similarity with the reference station(s) increases, values range from 0 to 1.0 for stations with no taxa in common to stations with same taxa as reference sites. Coefficient was calculated as $JC = a/(a + b + c)$ where a is the number of taxa in common to both reference and comparison sites, and b and c are the number of taxa unique to the comparison and reference sites. Index calculated at genus level because genera are more widely distributed in Idaho than species. For example, *Catostomus ardens* only occurs upstream of Shoshone Falls and *Catostomus columbianus* only occurs downstream; genus *Catostomus* is ubiquitous but not species within the genus. The five most frequently occurring genera at reference streams were used to define the assemblage. and because including rare species makes c large, and when c is large all JC values are low and stable, which provides no useful information.
Percent sculpins (Cottids) Mottled, Paiute, shorthead, Shoshone, slimy, torrent, and Wood River sculpins require well-oxygenated rubble or rubble/gravel substrate, and are absent or rare from streams with fine-grained substrates, highly embedded cobble substrates, or elevated metals (McCormick et al. 1994, Mebane 2001). Larvae of these species and some adults burrow into the interstitial spaces of cobble substrate for refuge (Bond 1963, Finger 1982, Haro and Brusven 1994). Some freshwater sculpin species (e.g. reticulate, Pit) are tolerant of fine sediments and low dissolved oxygen, but do not occur in the study area (Zaroban et al. 1999). Sculpin have similar physiological needs as many salmonids, but relatively sessile habits make them excellent water-quality indicators (Carline et al 1994, Bond 1963). Sculpin species have wide but disjunct distributions. They may be absent from small, high elevation, or high gradient reference streams. Sculpin diets overlap with trout, and when abundant, may reduce food consumption and production of salmonids (Johnson 1985, Brocksen et al. 1968). Percent sculpin and sculpin age-class metrics may be functionally similar.
Percent salmonids Several IBI models for cold water or mixed cool and cold water assemblages have used the percentage of salmonids as an indication of environmental suitability for intolerant species (Table 1; Barbour et al. 1999). This metric is conceptually similar with the percentage of cold water species metric, since almost all cold water species in the sample sites were salmonids and sculpins.

Table 4-2. *(continued)*

Indicator species metrics
Percent sensitive native individuals Tolerances of many species to environmental stress have been listed for many species (Zaroban et al. 1999). River systems that are similar to natural reference conditions will include sensitive native individuals. Conversely, sensitive natives will be the first to decline in a system that is highly turbid, silty, or warmer than historic conditions
Percent tolerant individuals The proportion of fishes that thrive in or tolerate poor-quality water is likely to be high in rivers with poor water quality. This metric includes some native cyprinids (minnows) and catostomids (suckers), as well as many introduced, non-indigenous species (e.g. carp, tench, bullheads, centrarchids).
Percent non-indigenous individuals This metric reflects the severity of biological pollution by dominance of invading or introduced non-indigenous species. The number of nonindigenous species is measured by how many species have become established, rather than dominance by any one or more non-indigenous species. Presence of a non-indigenous species is usually permanent and less variable than the percent of non-indigenous individuals.
Percent of cyprinids as longnose dace Overall, native cyprinids may increase their dominance in Idaho streams in response to degradation (Royer and Minshall 1996). However, of the non-cold water fishes (i.e. other than salmonids and sculpins), longnose dace's environmental requirements may make it a useful indicator species. They require well oxygenated, flowing water, and access to riffle habitats, where they deposit their eggs in rock crevices, and principle food items are dipterans and mayflies. The longnose dace seeks the interstices between stones in gravel-rock substrates of riffle areas of streams to spawn (Scott and Crossman 1998). Sedimentation, loss of riffle habitats, and diminished stream flows adversely affect longnose dace populations (Probst 1982, New Mexico Natural Heritage Program, http://nmnhp.unm.edu). Longnose dace are widely distributed in Idaho streams, and were frequently captured in rangeland streams (Table 4; Simpson and Wallace 1982). Probst (1982) reported longnose dace were affected by siltation of spawning riffles but otherwise tolerated a wide range of environmental conditions. Poff and Allan (1995) also reported longnose dace were sensitive to siltation in Minnesota and Wisconsin streams.
Trophic composition
Percent omnivore plus herbivore individuals This metric is a measurement of omnivore species that take significant quantities of both plant and animal materials (including detritus). It assumes that the dominance of omnivores occurs as specific components of the food base become less reliable in response to degradation. The opportunistic foraging habits of omnivores make them more successful than specialized foragers (Karr et al. 1986). This metric was broadened to include herbivores. Herbivores are considered specialized feeders, however they thrive when sunlight and nutrient conditions are sufficient to support lush plant growth. These conditions may result from removal of riparian shading and nutrient inputs from agriculture (Cuffney et al. 1997).
Percent insectivore individuals This metric assumes that as habitats are degraded, insect food sources are reduced in diversity, and there is a shift from insectivorous to omnivorous species (Karr et al. 1986). Most salmonids, all sculpin, and some native cyprinids are included in this metric. In the wadeable streams of this study area, carnivores, (i.e. piscivores-fish that feed on other fish) were uncommon. These species include northern pikeminnow (formerly known as the northern squawfish), smallmouth bass, and adult fluvial bull trout; these species are more common in large rivers and reservoirs. Thus, this metric is the inverse of, and redundant with, the *% Omnivores and Herbivores* metric.

Table 4-2. *(continued)*

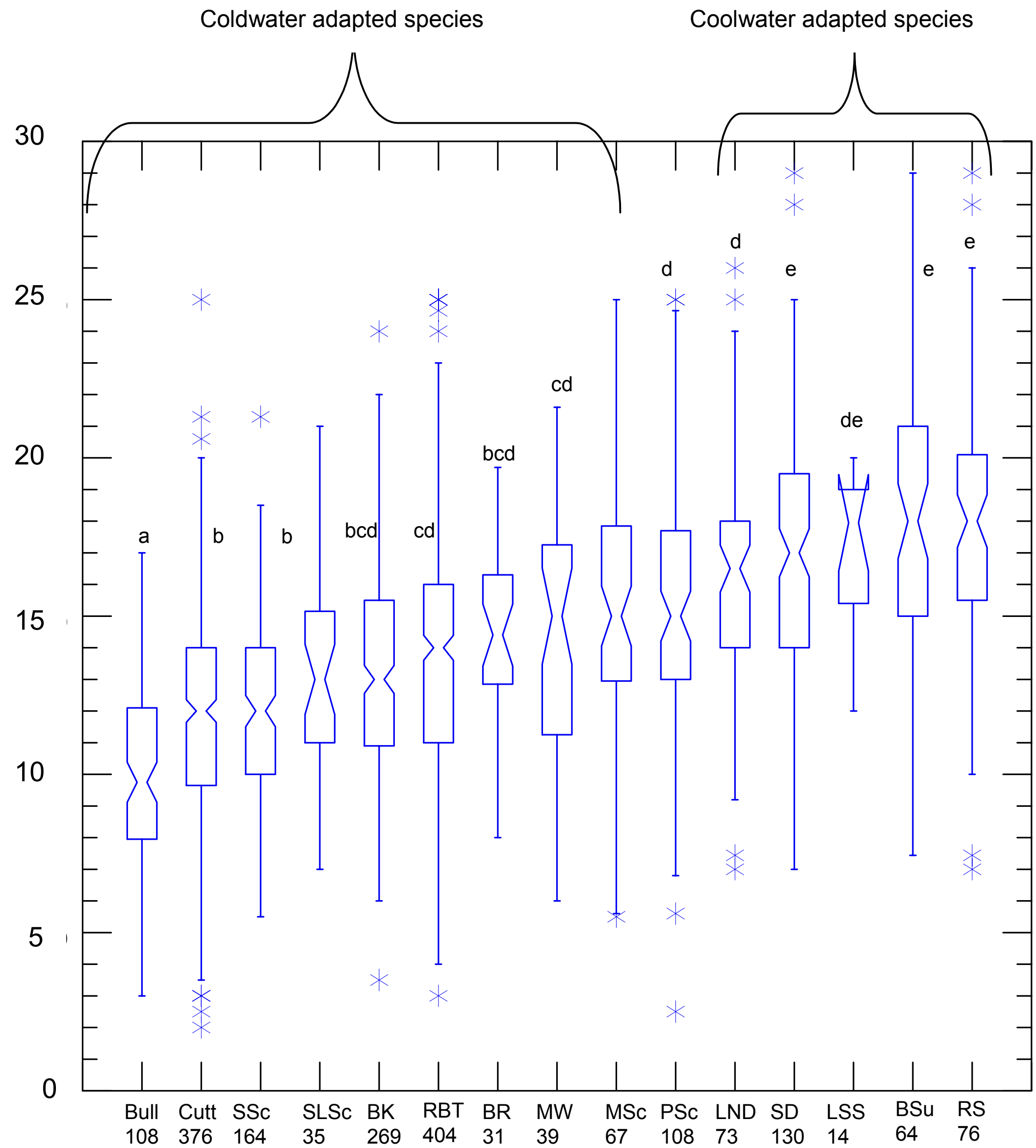
Reproductive function metrics
Number of salmonid age classes This metric reflects suitability and stability of conditions in a surveyed location for salmonid spawning, juvenile rearing, and adult salmonids. Age classes are inferred from measured size classes and typical length at age relationships (same for sculpin age class metrics). Shifts in age class distribution are frequently responses to different stressors, such as exploitation, recruitment failure, food limitation, or niche shifts (Munnkittrick and Dixon 1989).
Number of selected salmonid ("trout") age classes (Genus *Oncorhynchus*, *Salvelinus*, and *Salmo* [Trout, char, and salmon]; mountain whitefish are excluded.) This metric is similar in concept to salmonid age classes, but restricted to redd building gravel spawners. It excludes gravel or rock broadcast spawners with pelagic free embryos, which develop in river backwaters rather than in substrate interstices. It also excludes mountain whitefish due to their markedly different reproductive strategy from trout, salmon, and char. Mountain whitefish spawn by broadcasting eggs and milt over gravelly areas, and no redd is built (e.g. nonguarding open substratum, lithopelagophils, Simon (1998b)). Neither whitefish embyros or larvae appear to utilize intergravel spaces for either embyros or alevins; development occurs in quiescent river backwaters (Northcote and Ennis 1994). Trout bury their eggs in redds in gravelly areas (e.g. nonguarding brood hiding lithophils, (Simon 1999b)) The spawning success of trout may be more sensitive to siltation than whitefish because fines embedded in gravels reduces interstitial spaces between the gravel particles, in turn reducing intergravel dissolved oxygen necessary for egg and alevin survival. The surface layer of embedded gravels may still be suitable for broadcast spawners. The spawning success of trout and salmon are sensitive to excessive siltation because fines embedded in gravels reduces interstitial spaces between the gravel particles, in turn reducing intergravel dissolved oxygen necessary for egg and alevin survival (Chapman 1988, Maret et al. 1993). Trout age classes declined with increasing proportions of fine sediments in mountain streams in central Idaho (Mebane 2001). Mountain whitefish were abundant and multiple age classes were present at apparently degraded large river sites at which trout were absent or rare (Mebane 2000), and thus may be more resilient than other salmonids to temperature and low intergravel dissolved oxygen.
Number of sculpin age classes This number reflects the availability of un-embedded cobble substrate required for cavity nesters and juvenile refuge. Sedentary life histories result in adult home ranges of <50-150m (Hendricks 1997). Their low dispersal distances are advantageous for assessing site conditions over several years. Sculpin age classes declined with increasing proportions of fine sediments in mountain streams in central and eastern Idaho (Mebane 2001).

Table 4-2. *(continued)*

Abundance and condition metrics
Number of cold water individuals per minute of electrofishing (catch per unit effort, CPUE) Cold water fish should be more abundant at locations with favorable conditions for cold water biota. However a myriad of natural and anthropogenic factors that limit the abundance of fish complicates interpretation, particularly with trout that are subject to harvest. Erman (1986) reported that fish abundance was more variable than assemblage structure and was driven by climate changes. Defining this metric as cold water instead of salmonid abundance (Table 1) may lessen potential confounding harvest effects. Since abundances of all fish may increase in response to some types of degraded water quality (the paradox of enrichment), limiting the metric to cold water individuals may avoid that response. Estimates of density or biomass can be difficult to measure, for example sampling efficiency drops in larger waters and density estimates difficult in complex habitats (e.g. logjams). Abundance needs to be normalized to compare different-size habitats, different fishing efforts, etc. The DEQ sampling program typically uses single-pass electrofishing to sample the fish assemblage. Precise, unbiased abundance estimates may require a large number of removal passes or mark-recapture sampling. However, a single catch-per-effort pass will sample a consistent proportion of the fish species in small streams, in terms of both richness and abundance, when compared with removal sampling. Also, if the stream stations are about 35 times the mean stream width, the influence of fish entering and leaving the station may be negligible, making block nets unnecessary for CPUE sampling (Simonson and Lyons 1995). Kruse et al (1998) also compared single-pass sampling efficiency to intensive methods since intensive sampling to obtain unbiased population estimates limits the ability of field crews to assess large areas. Kruse et al. found that in trout streams with low densities (<0.4 fish/m2) and where most refuge habitat was in boulder pools, with limited amounts of instream vegetation, undercut banks, or woody debris, single pass electrofishing was sufficient for estimating fish abundance in small mountain streams. The stream features in Kruse et al.'s (1998) study area in the Middle Rockies ecoregion (Omernik and Gallant 1986) likely comparable to those in similar mountain ecoregions of Idaho.
Percent of fish with DELT anomalies Fish may develop external deformities, eroded fins, lesions, or tumors (DELT anomalies) in response to exposure to contaminated sediments or other exposure routes. This metric excludes protozoan or other parasitic infestations that are often unrelated to water quality. Issues: Numerous field and laboratory studies on fish have shown that the liver is the most frequent site of neoplasms in fish that were correlated with chemical contamination. Less common neoplasms of other epithelial tissues, including the skin, have been linked to environmental contamination. Etiology of other gross external pathological anomalies are inconsistent and have not been closely linked to chemical contamination (Harshbarger and Clark 1990, Horness et al. 1998). Yet, DELT anomalies may be more common in areas with significant urban agricultural development than at reference sites. In this study area, Munn and Gruber (1997) found more frequent detections of organochlorine compounds in fish tissue collected from agricultural streams than from streams without agriculture as a major upstream land use. Frequency of DELT anomalies was also higher at agricultural streams (Munn. unpub. data), but no correlations between concentrations, frequency of chemical detections, and frequency of DELT anomalies were found. Mebane (2000) found DELT anomalies were about twice as high at river sites which significant agricultural development upstream than at other sites, suggesting an association between pesticide use and anomalies. Sanders et al. (1999) qualitatively associated the occurrence of DELT anomalies to pollution sources in Ohio rivers. While DELT anomalies are a crude measure, with frequent false positive results (i.e. DELT anomalies without chemical stressors) , the possibility of identifying areas with elevated incidences of cancer or other abnormalities in wild fish for further study outweighs these limitations.
Percent of fish with any anomalies This metric is the same concept as with DELT anomalies, but not limited to deformities, eroded fins, lesions, or tumors in response to exposure to contaminated sediments or other exposure routes. It includes protozoan or other parasitic infestations, such as blackspot disease, which are often unrelated to water chemistry.

Cold water species evaluation

Accurate classification of fish species attributes is a fundamental requirement that must be met before an IBI can be developed. These attributes include thermal tolerance, tolerance of low dissolved oxygen and siltation, and trophic guild. Zaroban et al. (1999) recently published a comprehensive list of these attributes for 132 Pacific Northwest species. The authors cautioned that because of the breadth of their compilation, their classifications were necessarily subjective, and based primarily on previous reviews (e.g. the state "Fishes of..." books). Testing most of the classifications would be major studies unto themselves, and would certainly be beyond the scope of this analysis. However, the regional temperature classifications can be evaluated and potentially refined with more geographically specific (i.e. Idaho) matched temperature and species occurrence records. Temperature measurements were usually made at the beginning of each electrofishing pass. Therefore, each fish was captured within about 20 minutes and 100m of that single point temperature measurement. While stream temperatures are variable, even within 20 minutes and 100m, it is reasonable to assume that each fish captured could have experienced the instantaneous temperature measured at the beginning of each electrofishing pass. Singly, this means little; but when the distributions from many such records are plotted, patterns of temperature preference, overlap, and separation between species begin to take shape (Figure 4-2).



Code	Common Name	Scientific Name	Code	Common Name	Scientific Name
Bull	Bull trout	*Salvelinus confluentus*	MW	Mountain whitefish	*Prosopium williamsoni*
SSc	Shorthead sculpin	*Cottus confusus*	MSC	Mottled sculpin	*Cottus bairdi*
Cutt	Cutthroat trout	*Oncorhynchus clarki*	PSC	Paiute sculpin	*Cottus beldingi*
SLSC	Slimy sculpin	*Cottus cognatus*	LND	Longnose dace	*Rhinichthys cataractae*
BK	Brook trout	*Salvelinus fontinalis*	SD	Speckled dace	*Rhinichthys osculus*
RBT	Rainbow trout	*Oncorhynchus mykiss*	LSS	Largescale sucker	*Catostomus macrocheilus*
BR	Brown trout	*Salmo trutta*	BSU	Bridgelip sucker	*Catostomus columbianus*
			RS	Redside shiner	*Richardsonius balteatus*

Figure 4-2. Temperatures at which selected fish species were captured by electrofishing in Idaho from 1994-1998. Plots are in rank order by median temperatures. The boxes indicate the median and upper and lower quartiles (the central 50% of the values), the whiskers extend up to 1.5X the interquartile value, and asterisks show outlying values. Species with few matched temperature and capture records were excluded. Plots marked with the same letter indicates that their means are not significantly different at $P < 0.05$ using Tukey's multiple comparison procedure.

The distribution of temperature at which selected fish species were captured supports the division of the fish species into broad categories of mesothermal (cool water adapted) cyprinid-catostomid species groups and stenothermal (cold water adapted) cottid-salmonid species groups. Within the stenothermal group, species can be grouped as moderately stenothermal (rainbow trout, mountain whitefish, mottled and Paiute sculpin) and strongly stenothermal groups (cutthroat and brook trout, shorthead and slimy sculpin).

The temperature distributions for bull trout support the viewpoint that bull trout are extraordinarily stenothermal. In our summertime surveys, bull trout were found in both the coldest waters (9.8°C median temperature), and among the most limited range of temperatures for the stenotherms. Most bull trout were found at temperatures below 12°C (75th percentile of temperatures) and were never captured above 17°C.

These field data support the qualitative thermal classifications listed in Zaroban et al. (1999). Median stream temperatures at which the stenothermal, cold water-classified fishes were collected were approximately less than 15°C. The sole difference suggested between these distributions and their list is that regionally mottled sculpin were listed as cool water. The data shown in Figure 4-2 suggest that the cool water-classified mottled sculpin, cold water-classified Paiute sculpin, and cold water-classified mountain whitefish have similar summertime temperature occurrences. Mottled sculpin are included in the cold water metrics used here (e.g. cold water native species, percent cold water species).

<u>Native species evaluation</u>

Several of the metrics are defined as native or cold water native species. For most species, determining whether a species captured at a given stream is or is not indigenous is a simple matter of consulting a reference list (e.g. Simpson and Wallace 1982, Zaroban et al. 1999). However, in the case of the *Oncorhynchus mykiss*, which may be considered either a rainbow, redband, or steelhead trout, it is not a simple matter to determine whether a population is native or introduced. Behnke (1992) states that only two forms of *O. mykiss* are native to Idaho, the interior redband trout and the anadromous steelhead trout. Either or both or both forms were historically widely distributed in the lower Snake River and its tributaries as far upstream as Shoshone Falls (Figure 3). Subsequently, the coastal rainbow trout form of *O. mykiss* has been extensively stocked within and beyond the historical range of *O. mykiss* in Idaho. Determining whether an individual *O. mykiss* was a wild fish, or resulted from natural reproduction by a previous introduction is very difficult. Thus, for this report, unless an *O. mykiss* individual could be identified as a hatchery fish by its appearance, any *O. mykiss* caught within their historical range (Figure 4-3) is presumed to represent a native species at that site. *O. mykiss* captured beyond their historical distribution are always scored as non-indigenous species in the “native species” metrics (e.g. upper Snake River basin, Spokane River basin).

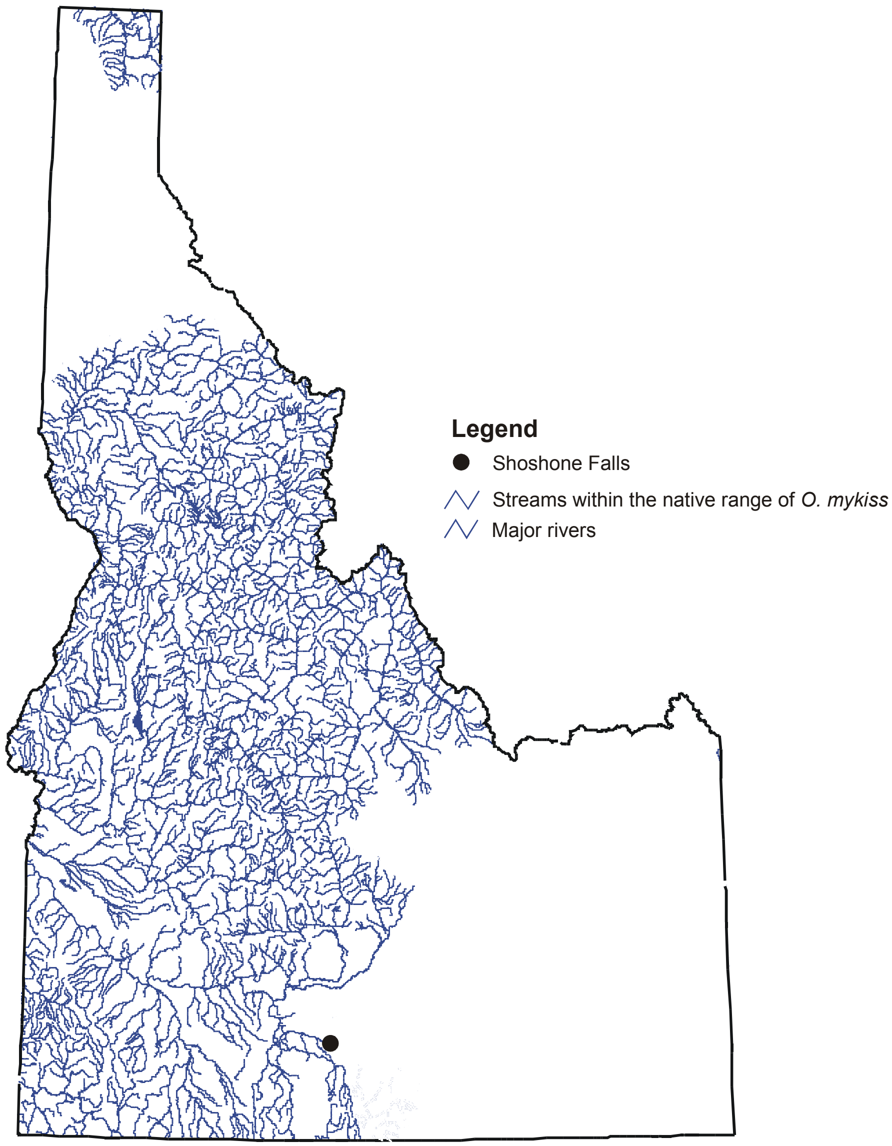


Figure 4-3. Native distributions of *Oncorhynchus mykiss* (steelhead, redband, and rainbow trout) in Idaho. Historically, *O. mykiss* occurred only in the Snake River basin downstream of Shoshone Falls and the Kootenai basin. *O. mykiss* found within their historical range are considered a native species, and non-native elsewhere. Historic distributions are from Behnke (1992).

Evaluation of anthropogenic disturbance

The utility of any bioassessment index depends on its ability to adequately respond to disturbance. Each candidate metric was a candidate because of assumptions that each measure of ecological structure or function would change in response to anthropogenic disturbance. This assumption needed to be tested for each metric, and for the combination of the metrics into an index, for the approach to be valid. This required some evaluation of disturbance for each sample location. Because no single quantitative measure of disturbance was available for all sites, a qualitative ranking scheme was used to group sites into apparent reference sites (minimally disturbed) and apparent test sites (apparently anthropogenically disturbed). Indeterminant sites were considered "other" and were not used in index development. The BURP sites were grouped by interviewing regional biomonitoring team leaders using the criteria in Table 4-3. The IDFG surveys included a similar field rating of observed land use proximate to the survey site (e.g. no use; no use observed [but the sampling crew was not certain there was in fact no ongoing use]; light grazing; heavy grazing; agricultural activities next to streams; etc. [K. Meyer, IDFG, Resident Species Research Program, pers. comm.]) IDFG sites coded as no use, no use observed, or light grazing were considered reference; agricultural or heavy grazing sites were considered test sites (the IDFG sites used were from rangeland streams with grazing and the primary human land use). Robinson and Minshall (1995) were studying characteristics of least-disturbed streams, so all of their streams were specifically selected as reference streams. NAWQA sites were assigned to reference or test groups following discussions with the study team leaders.

Table 4-3. Factors considered for classifying reference sites.

Variable	Criteria
Roads	Not constraining riparian zone, crossings are infrequent, no evidence of road associated failures from culverts or gullies to streams.
Riparian vegetation extensive and old	Riparian growth is considered extensive when it occurs all along the shoreline and is capable of shading the stream and buffering human influences. It is considered old when overhangs the stream or deposits large woody debris
Riparian structure complex	Complexity characterized by presence of a canopy, understory and groundcover (trees, shrubs, and groundcover
Channel complex	Mixture of pool, glide, riffle, and run habitat types
Habitat structure complex	Substrate heterogeneous
Chemical stressors likely minimal	Likely sources of chemical stress are few (e.g. unbuffered croplands, irrigation returns, active or in-active mining areas, regulated discharges), or if potential sources present, chemical data shows standards or guidelines met, and thus effects are unlikely.
Shoreline/channel modification minimal	Evidence of riprap, channel straightening, vegetation removal or other disturbances absent or minimal.
Flow modifications minimal	Upstream impoundments absent. Irrigation withdrawal or other diversions absent, or if present, likely cause minimal disruption to the hydrologic cycle (i.e. acknowledging that almost all streams located in the semi-arid basin/lowland ecoregions will have some water withdrawals)
Evidence of excessive sedimentation absent	Apparent anthropogenic sediment increases not noted (e.g. crop or road gullies, livestock bank trampling, mass wasting) No field notes of highly turbid conditions. No indications from habitat variables of excessive sedimentation (e.g. No "poor" qualitative cobble embeddedness estimates (≥75%), channel substrate <50% fine sediments (measured as bankfull).
Grazing in riparian zone minimal	Absence of laid back, trampled, or unstable banks.
Logging, construction, or other disturbances minimal	If present, buffered from riparian zone
Agricultural disturbances	Croplands not impinging riparian zone, runoff or irrigation returns minimal

Modified from Hughes (1995)

RESULTS

Classification of Stream Types

Discriminant analysis of species' abundances and presence-and-absence was used as a way to test for multivariate differences between the descriptive grouping of streams by ecoregion and elevation into the "rangeland" and "forest" stream types. The goal of discriminant analysis is to find a linear combination of variables that can be used to separate groups, or to assign new sites based on the model developed for known sites. Discriminant analysis is usually better at confirming group memberships for known samples than predicting group memberships of unknown samples (Matthews 1993). A limitation to the technique is that since it is constrained to linear combinations of variables, interpretation of nonlinear relationships is difficult (Engelman1996; James and McCulloch 1990). Here, the assumption was that ecoregions were natural aquatic ecosystem groupings, and that species' abundances, or presence-and-absence at reference sites would be separated by ecoregions.

Fish species' abundance was not differentiated by mapped ecoregions (Figure 4-4a). When plotted, confidence ellipses centered on the centroids of each groups' canonical scores overlapped almost completely. The differentiation was improved by grouping the ecoregions into elevation-adjusted aggregates of the predominantly "rangeland" and "forest" type ecoregion groups (Figure 4-4b). To see if reducing the noise in the abundance data improved the separation, the canonical scores were plotted after tranforming the fish species occurrence into simple presence-and-absence data, ignoring the abundances or abundance ranks of the species (Figure 4-4c). The presence-and-absence and species' abundance plots were similar; group differentiations were somewhat stronger based upon species abundance plots.

The pool of shared species between ecoregions is likely a factor in the large overlap in the rangeland and forest groups. Of the most frequently occurring species in both ecoregion groups, five are common to both. Only the rank of their abundance differs (Table 4-4). Rainbow trout were the most common species encountered in both groups. The results of these analyses suggest that for the purpose of grouping fish assemblages based upon species composition, these elevation-adjusted ecoregion groups improved upon ecoregions as a differentiator. However, their is much overlap between the species composition of these ecoregion groups, and therefore, while these "rangeland" and "forest" stream distinctions may be intuitive and useful, they are by no means hard and fast.

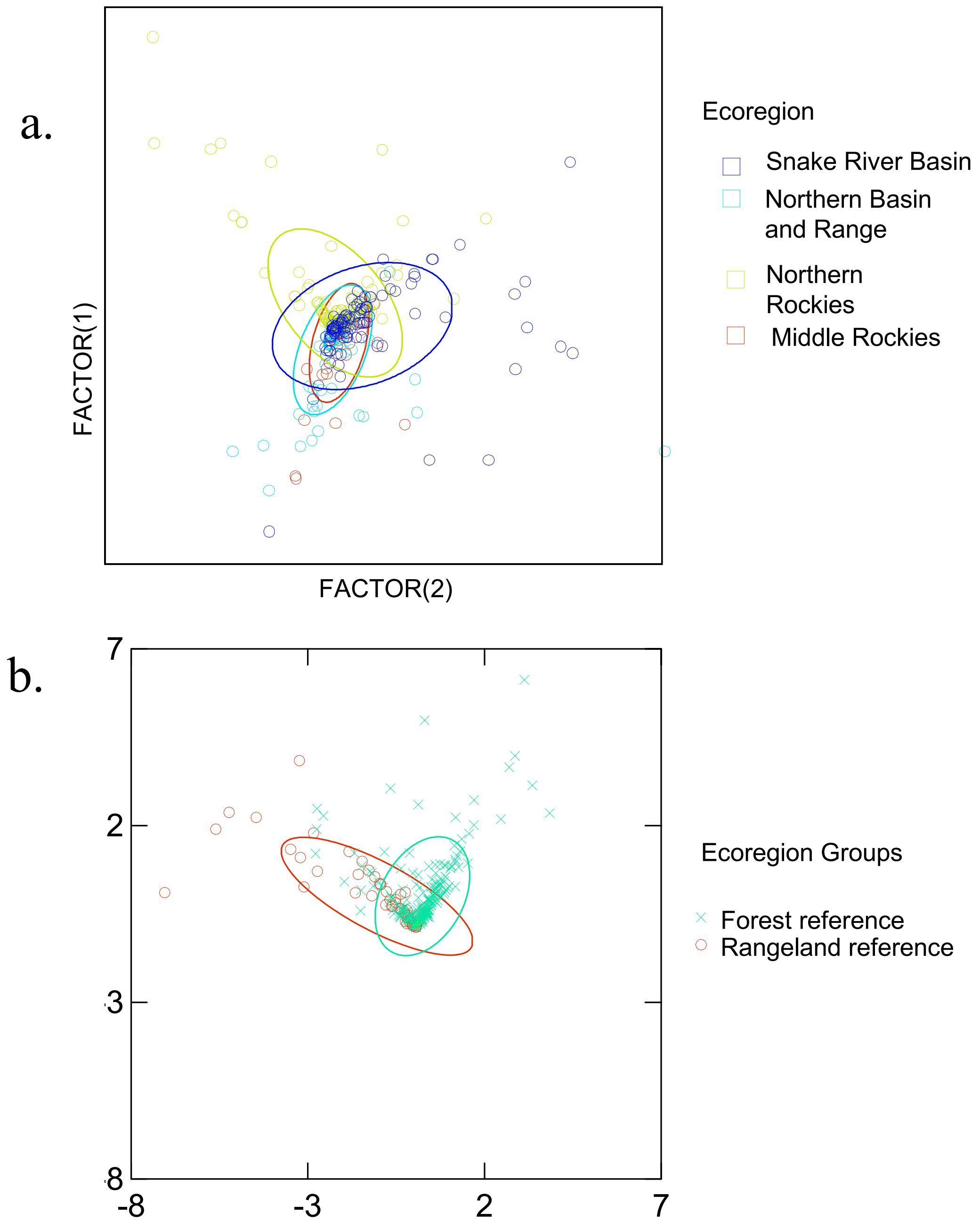


Figure 4-4. Discriminant analysis of reference site species occurrence and abundance grouped by ecoregions (a, top), and elevation-adjusted ecoregion aggregates (b, bottom). Groups clusters nearly completely overlap, indicating poor discrimination between ecoregions.

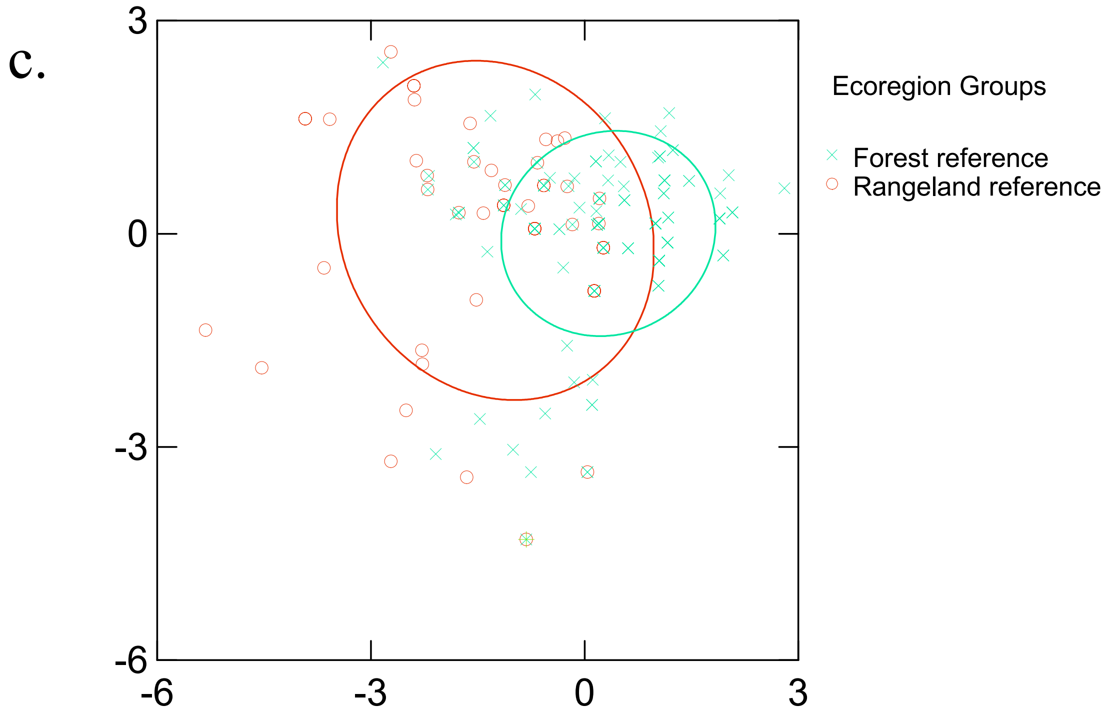


Figure 4-4. *(continued)* Discriminant analysis of reference site fish species presence-absence (12 most common species with a frequency of occurrence of at 0.1 in either group) grouped by elevation-adjusted ecoregion aggregates.

Table 4-4. Most frequently occurring species and genera in reference sites in forest and rangeland stream types.

Ranking by species and frequency of occurrence (%)									
Forest streams									
1st	2nd	3rd	4th	5th	6th	7th	8th	9th	10th
Rainbow trout	Cutthroat trout	Brook trout	Shorthead sculpin	Bull trout	Mottled sculpin	Paiute sculpin	Chinook salmon	Specked dace	Redside shiner
36%	30%	26%	26%	20%	11%	9%	7%	7%	6%
Rangeland streams									
Rainbow trout	Speckled dace	Redside shiner	Mottled sculpin	Cutthroat trout	Longnose dace	Mountain sucker	Paiute sculpin	Bridgelip sucker	Brook trout
69%	38%	31%	25%	18%	16%	15%	14%	11%	8%
Ranking by genus and frequency of occurrence (%)									
Forest streams									
1st	2nd	3rd	4th	5th	6th	7th	8th	9th	10th
Oncor.	*Cottus*	*Salve.*	*Rhini.*	*Richard.*	*Cato.*	*Prosop.*	*Salmo*	*Micro.*	*Ptycho.*
77%	53%	46%	12%	5.7%	5.7%	2.5%	1.6%	0.8%	0.4%
Rangeland streams									
Oncor.	*Rhini.*	*Cottus*	*Cato.*	*Richard.*	*Salve.*	*Prosopium*	*Ptycho.*	*Salmo*	*Micro.*
89%	58%	46%	34%	31%	9.2%	6.2%	6.2%	1.5%	1.5%

Genus abbreviations:
Oncor. – *Oncorhynchus* (Pacific trout and salmon)
Cottus – *Cottus* (sculpin)
Salve. – *Salvelinus* (char trout)
Richard – *Richardsonius* (shiners)
Rhini. – *Rhinichthys* (dace)
Cato. – *Catostomous* (suckers)
Prosop. – *Prosopium* (whitefishes)
Salmo . – *Salmo* (Atlantic trout and salmon)
Ptycho. – *Ptychocheilus* (pikeminnows)
Micro. – *Micropterus* (bass)

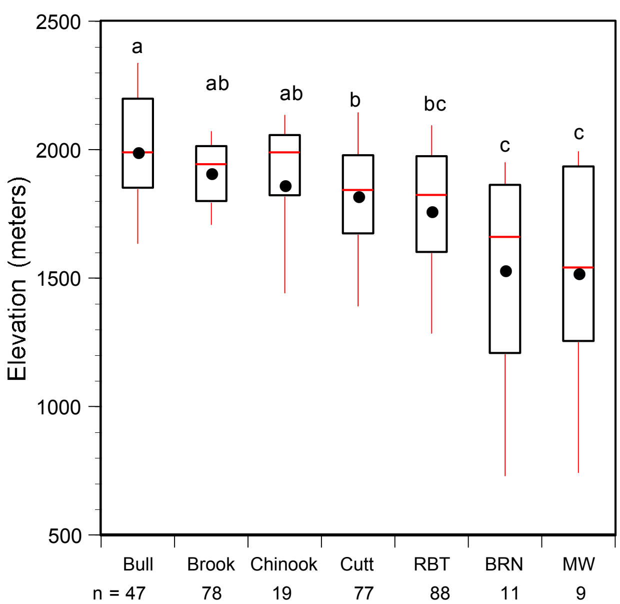
Forest Streams

Results in the “forest,” or “mountain” ecoregion groups, were considered with the expectation that within this ecoregion group, elevation and stream size were additional factors influencing the natural stream assemblage of fishes (e.g. Platts 1979, Vannote et al 1980, Li et al. 1987). Further, an expression of anthropogenic disturbance to stream habitats may be that the natural elevational zonation of fish species in a stream can be shifted upstream (Rahel and Hubert 1991). Some comparisons of effects of elevation and stream size on fish assemblages from reference streams are discussed below, followed by metric testing and index construction.

Effects of elevation and stream size

Distribution of salmonid species by elevation – Seven salmonid species were captured frequently enough in the reference sites to make some observations about their occurrence at different elevations. In order of average occurrence by decreasing elevation, the salmonids were distributed as follows (Figure 4-5):

> Bull trout > brook trout > chinook salmon > cutthroat trout > rainbow trout > brown trout > mountain whitefish .



Data limited to stations located between latitude 43° 30' and 45° 30' to lessen the effects of altitude. Dot indicates the mean. For each species marked with the same leter, the means were not significantly different (P<0.05), using Tukey’s multiple comparison procedure.

Figure 4-5. Elevations at which salmonid species occurred in wadeable streams (1st to 4th order).

These rankings are similar to, but not identical to, the rankings by cold water occurrence (Figure 4-2). The relative ranks of brook trout and cutthroat trout changes, for example. One difference between the comparisons is that the rankings by elevation were restricted to reference sites only, whereas the rankings by temperature were for all sites with matched temperature and capture data. The purpose of the comparison was to see if the distributions suggested an additional elevation stratification was needed. While there are elevational patterns, particularly with bull trout and rainbow trout, there was significant overlap in the elevations at which the common salmonid species were captured. This overlap suggests that additional stratification by elevation within the forest ecoregion groups may not be that helpful for bioassessment.

Distribution of salmonids and sculpins by elevation, stream size, and gradient – Salmonids and sculpin are ubiquitous in mountain streams, and were usually the only families of fish encountered in this analysis. Sculpins may be a useful indicator of chronic or episodic pollution and may decline or be extirpated from streams in which native salmonids remain present or even abundant (Carline et al 1994, McCormick et al. 1994, T.R. Maret personal communication). Some sculpin species appear to have similar physiological needs trout, but their relatively sessile habits may make sculpin slower to recolonize episodically unsuitable habitats than the more motile salmonids (Bond 1963, Carline et al. 1994). However, the relative distribution of sculpins and salmonids in mountain reference streams varies greatly, and is apparently unrelated to environmental impacts. In many streams, only salmonids occur, particularly in the smaller streams. Thus, the challenge is to distinguish when the scarcity of sculpin may indicate an episodic disturbance from natural conditions where only salmonids occur and sculpin are absent or sparse.

Trout and sculpin relative abundances were plotted to consider whether the "trout-sculpin" zone could be divided into a "trout" and a "trout-sculpin" zone (e.g. McPhee 1966, Rahel and Hubert 1991). To avoid confounding the effects of disturbance, latitude and elevation on temperature, data were limited to reference streams within two degrees of latitude (43° 30'N to 45° 30'N). In small streams, sculpin became less common above about 2000m and absent above 2300m. This is consistent with Maret et al.'s (1997) observations that sculpin were rarely encountered above 2000m in small streams in the upper Snake River Basin. However, in the larger 3rd and 4th order streams, there was no obvious relationship between salmonid-sculpin distributions and elevation; the highest large stream sampled at about 2400m had about even abundances of trout and sculpin (Figure 4-6). These results did not suggest that additional elevational stratification within the forest-mountain stream ecoregion groups would be useful, except perhaps for small streams only. As the bottom graph in Figure 4-6 shows, overall, the median percent composition of small reference streams was 100 percent salmonids, whereas for the 3rd and 4th order reference streams, 75 percent had less than 100 percent salmonids. Thus, stream size could be used as stratifier between trout streams and trout-sculpin streams.

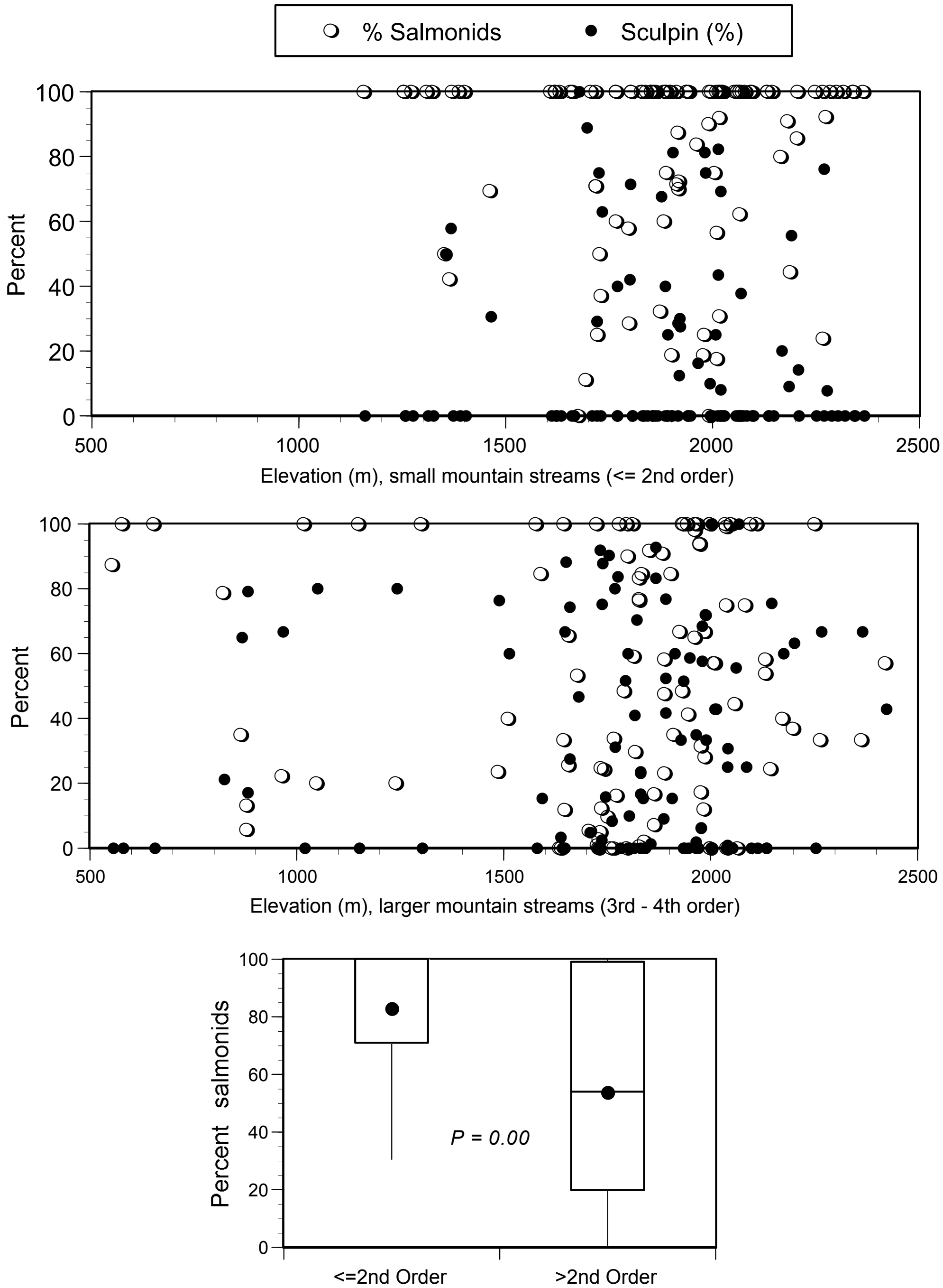


Figure 4-6. Relative abundance of salmonids and sculpin in mountain reference streams. Streams with salmonids comprising the entire fish assemblage are common, particularly in the small mountain streams.

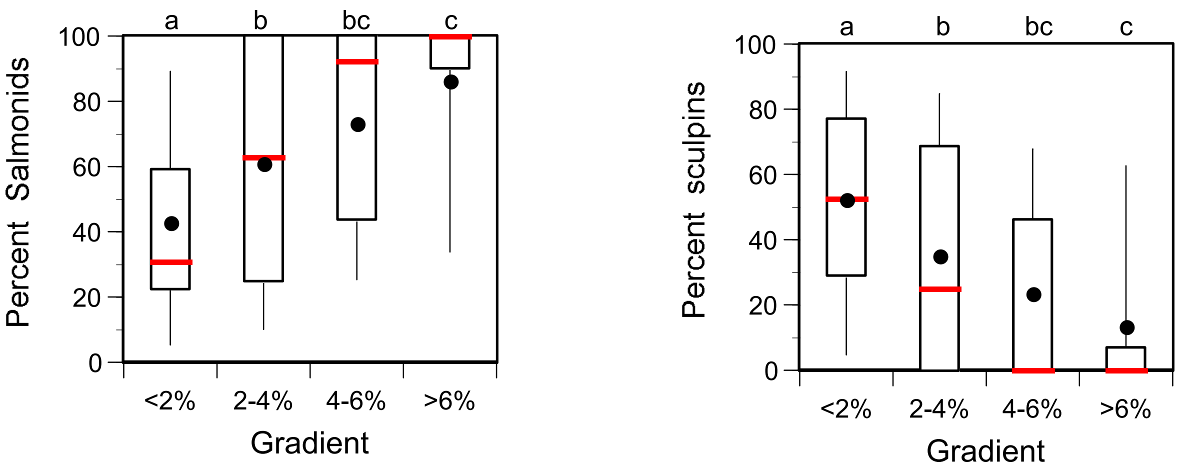
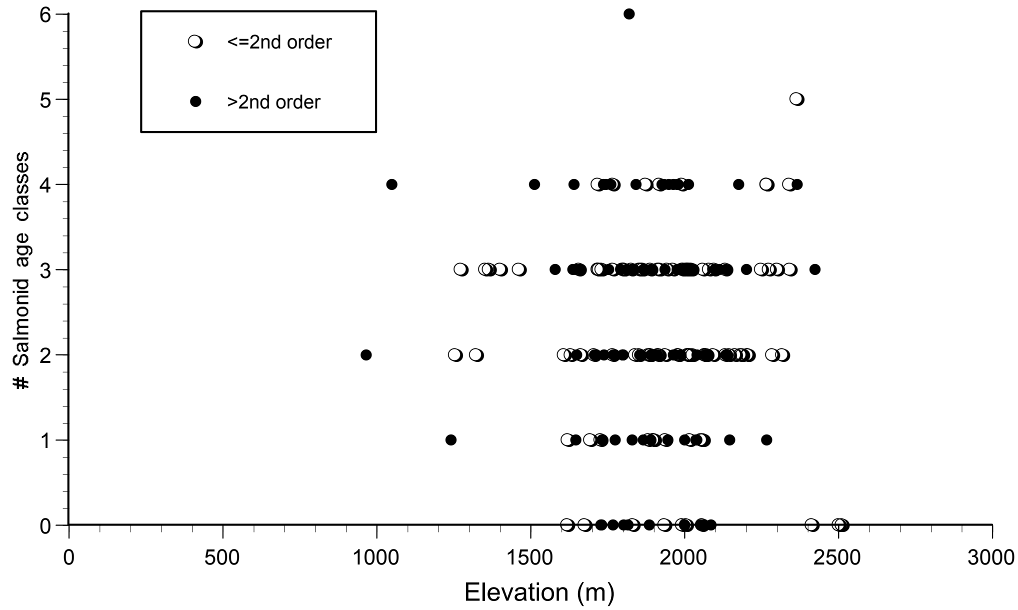


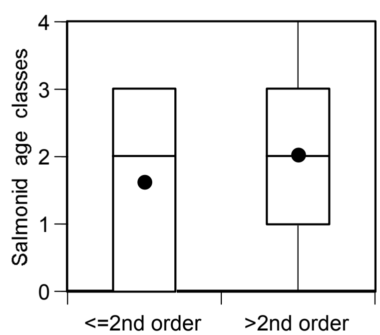
Figure 4-7. Salmonid - sculpin composition with gradient in forest reference streams. The means of groups marked with the same letter are not significantly different at P<0.05.

The relative composition of salmonids and sculpins in forest streams showed a clear graded change with stream gradient (Figure 4-7). At low gradient sites (less than 2 percent), sculpins were usually the most abundant fish captured, averaging over 50 percent of the assemblage. As stream gradient steepened, the proportions of salmonids increased and the proportions of sculpins decreased. At stream gradients greater than 4 percent , the median proportion of salmonids captured was greater than 90 percent, and the median proportion of sculpins was 0 percent .

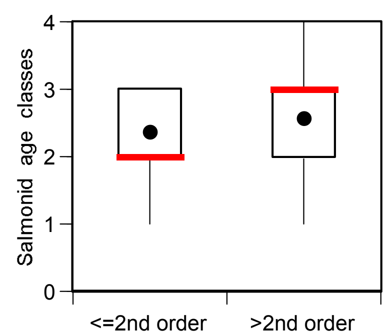
Gradient, stream size, and elevation are inter-related variables. Streams at high elevations are usually low stream order, small, and steeply cascading. Gradient appears to best explain natural distribution patterns among the salmonids and sculpins. Since the salmonids and sculpins together make up the vast majority of fishes in these mountain-forest stream types (Table 4-4, Figure 4-7), stream gradient needs to be factored into the stream classification in order to interpret biosurveys of these stream types.



Number of salmonid age classes captured at small and larger mountain reference streams (small stream ≤ 2nd order > larger stream) by elevation.



Salmonid age classes by reference stream sizes



Salmonid age classes, excluding fishless streams

Figure 4-8. Relationships between stream size and salmonid age classes encountered. In the box plots, age classes are shown for all mountain reference streams, including those with no fish, and limited to those reference streams with fish. Where medians overlap with other quartiles, medians are the thicker lines.

Salmonid age class presence – Whether age classes were more limited at upstream sites was tested. Summertime age classes of salmonids were segregated by stream size, and tested whether older and larger fishes occupying higher order, lower elevation streams, and juvenile fish in the headwater streams. The data did not give clear support to this idea (Figure 4-8). Except at the highest elevations sampled, where no fish occurred, no relationship between elevation and salmonid age class expression was apparent. Fish were rarely found above 2400m. There was an apparent relationship between stream size and salmonid age class strength (Figure 4-8, box plots). However, when excluding reference streams with no fish (presumably fishless because of flows, barriers, gradient or other non-anthropogenic effects), the median number of salmonid age classes captured at larger stream sites was higher than at small stream sites (3 vs. 2).

Abundance of cold water fishes – The relative abundance of cold water-adapted fishes was similarly compared by elevation and stream size (Figure 4-9). Relative abundance of cold water fish was measured through the number of cold water individuals captured per minute of electrofishing (catch-per-unit-effort, CPUE). At moderate elevations, between 1600m and 2200m in central Idaho, no relationship between elevation and cold water fish abundance was apparent. Statistical differences were tested for after grouping streams fish by size (≤ 2nd order and ≥3rd order) and then into groups by 250m elevations (<1750m, 1751-2000m, etc. CPUE was not statistically significantly different within these elevations.

When limiting the evaluations of abundance versus stream size to only those streams with fish, the box plots suggest a slight pattern of higher abundances occurring in the larger streams. The differences were not statistically significant (P=0.3, Mann-Whitney).

Since some data sets did not list their electrofishing effort, but areal density was reported or could be calculated, I looked at the relationship between the two measures (Figure 4-12). The two measures were related well enough that either could be used in an index, or one roughly estimated from the other. The purpose of including abundance in an index of biological integrity is to draw an inference about the relative water and habitat quality, rather than make absolute population estimates for fisheries management purposes. Regression errors from predicting one value from the other were largest at the highest values; metric scores based upon mid-range values could use areal density to estimate CPUE as follows:

$$\text{CPUE} \approx 18^{*}\ (\text{number cold water fish captured/square meter}) \qquad r^2 = 0.48$$

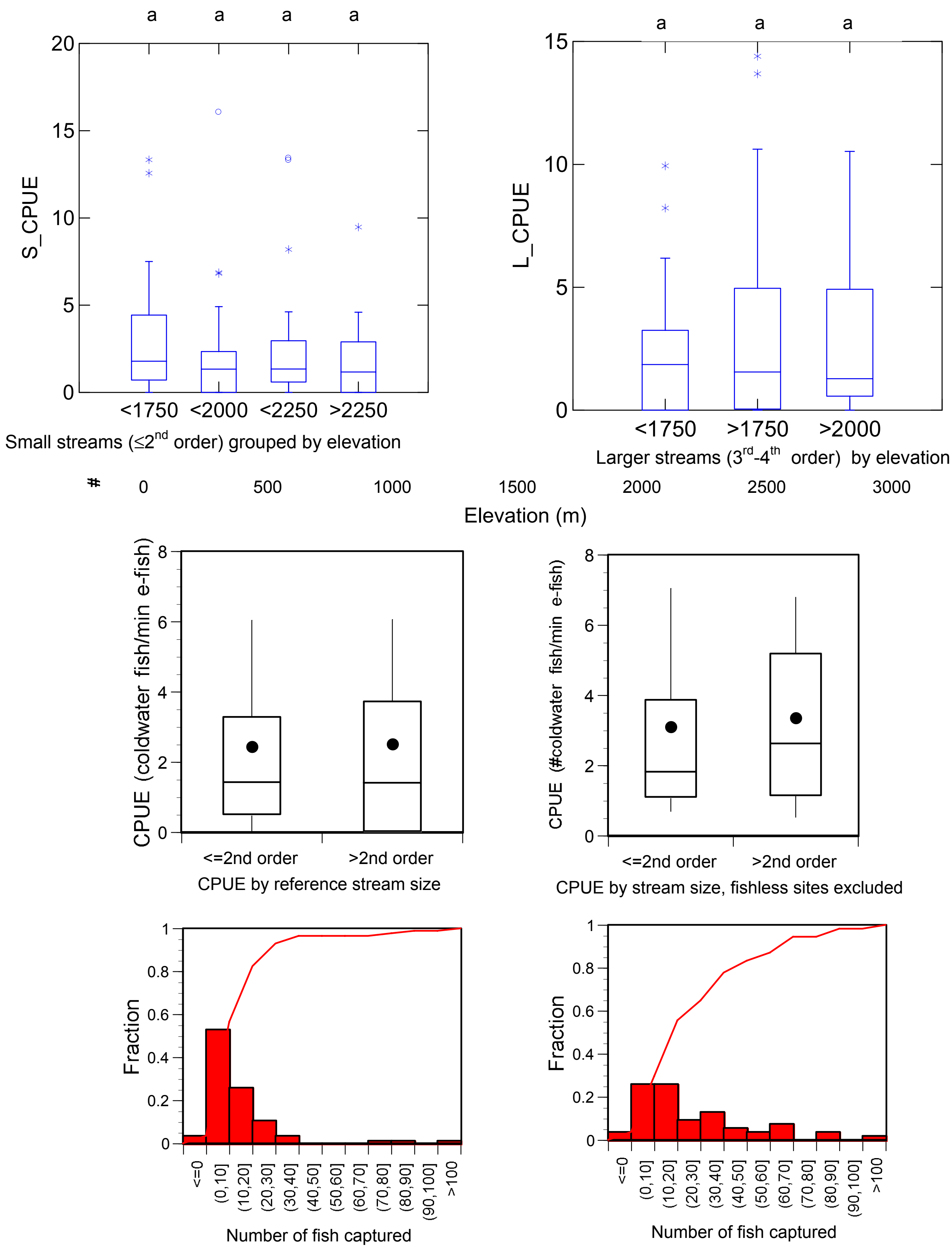


Figure 4-9. Elevation, stream size, and catch per unit effort (CPUE as # cold water fishes/minute electrofishing). In the upper box plot, groups marked with the same letter were significantly different ($P<0.05$). In the center box plots, CPUE is shown for all mountain reference streams, including those with no fish (left), and limited to those reference streams with fish (right).

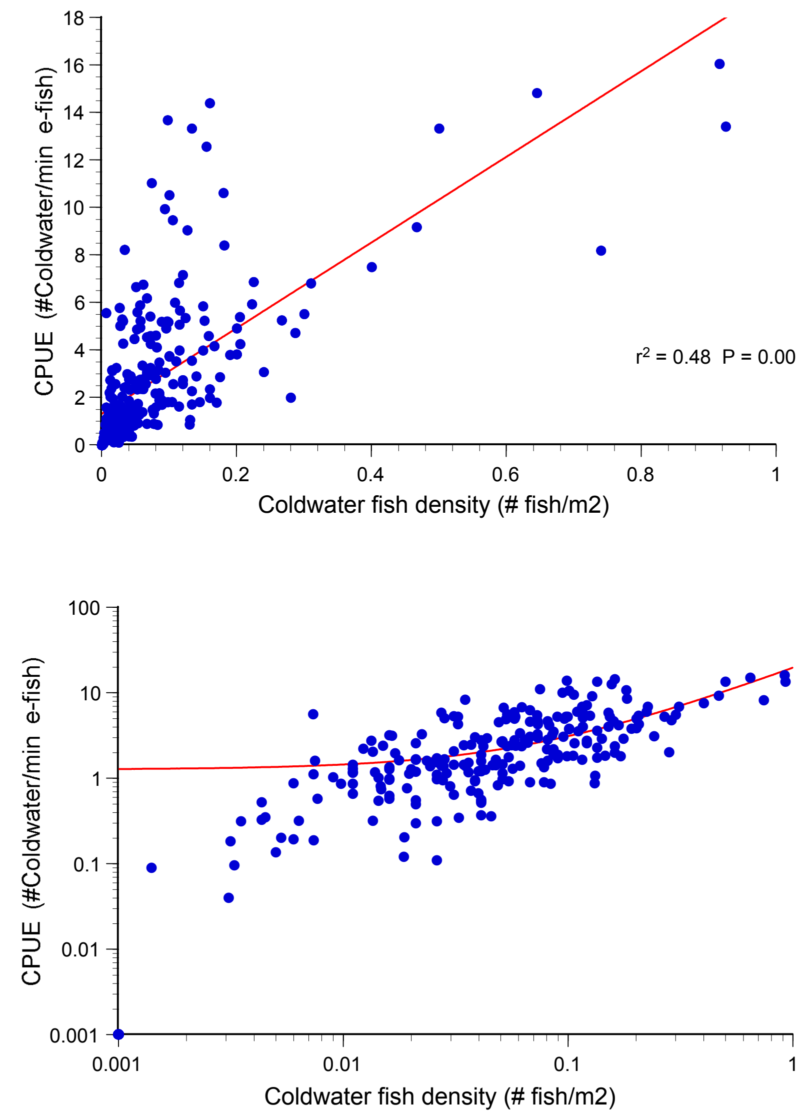


Figure 4-10. Relative densities of cold water fish were strongly correlated with catch per unit effort (CPUE), expressed as the number of cold water fish captured/minute of electrofishing. The plots are identical except that the lower plot uses a log-linear scale to spread out the data for better visibility. Comparisons limited to 1st pass of the reach, to keep data comparable between sites.

Metric evaluation and scoring for forest streams

After examining the influence of natural factors (elevation, stream size, gradient) on some potential metrics, metric values were compared at reference and test (potentially disturbed) streams (Figures 4-11 and 4-12). Six metrics that responded as predicted (i.e. higher values at reference sites) were included in the forest stream fish index (SFI):

1) # of cold water native species,

2) % cold water individuals,

3) % sensitive native individuals,

4) # of sculpin age classes,

5) # of selected salmonid age classes, and

6) relative abundance (number of cold water individuals/minute of electrofishing)

Metrics that appeared to have different ranges of values for small or medium sized streams were scored separately (Figure 4-15). Rationale for their inclusion and scoring scoring considerations follow.

Number of cold water native species – In the forest (mountain) streams, species richnesses were very low, usually only one or two species in reference headwaters streams, and one to three in intermediate-sized reference streams (Figures 4-11 and 4-12). Despite these limited ranges of values, there were clear distinctions between reference and test streams, and the absence of *any* native cold water species should be biologically meaningful. In small forest streams, presence of at least one cold water native species is scored as 1.0, since it was common for small reference streams to only have one cold water native species present (Figure 4-13). In larger forest streams, two cold water native species were expected and were scored as 1.0.

Percent cold water individuals – Reference streams in the forest ecoregion groups were usually occupied by nearly all cold water individuals with a median of 100 percent cold water adapted individuals and a mean of greater than 90 percent (Figures 4-11 and 4-12). Cold water salmon, trout and sculpins may directly compete for food or space with cyprinids such as redside shiner and speckled dace. These competitions are temperature mediated, with the cold water fishes dominating the cool water cyprinids at low temperatures (≈ 12 - 15°C) or sculpin have a competitive advantage over cool water cyprinids (Baltz et al. 1982; Reeves et al 1985, Hillman 1991, Taniguchi et al 1998). The metric was scored as a curve with greater than 90 percent cold water individuals receiving the highest scores (1.0).

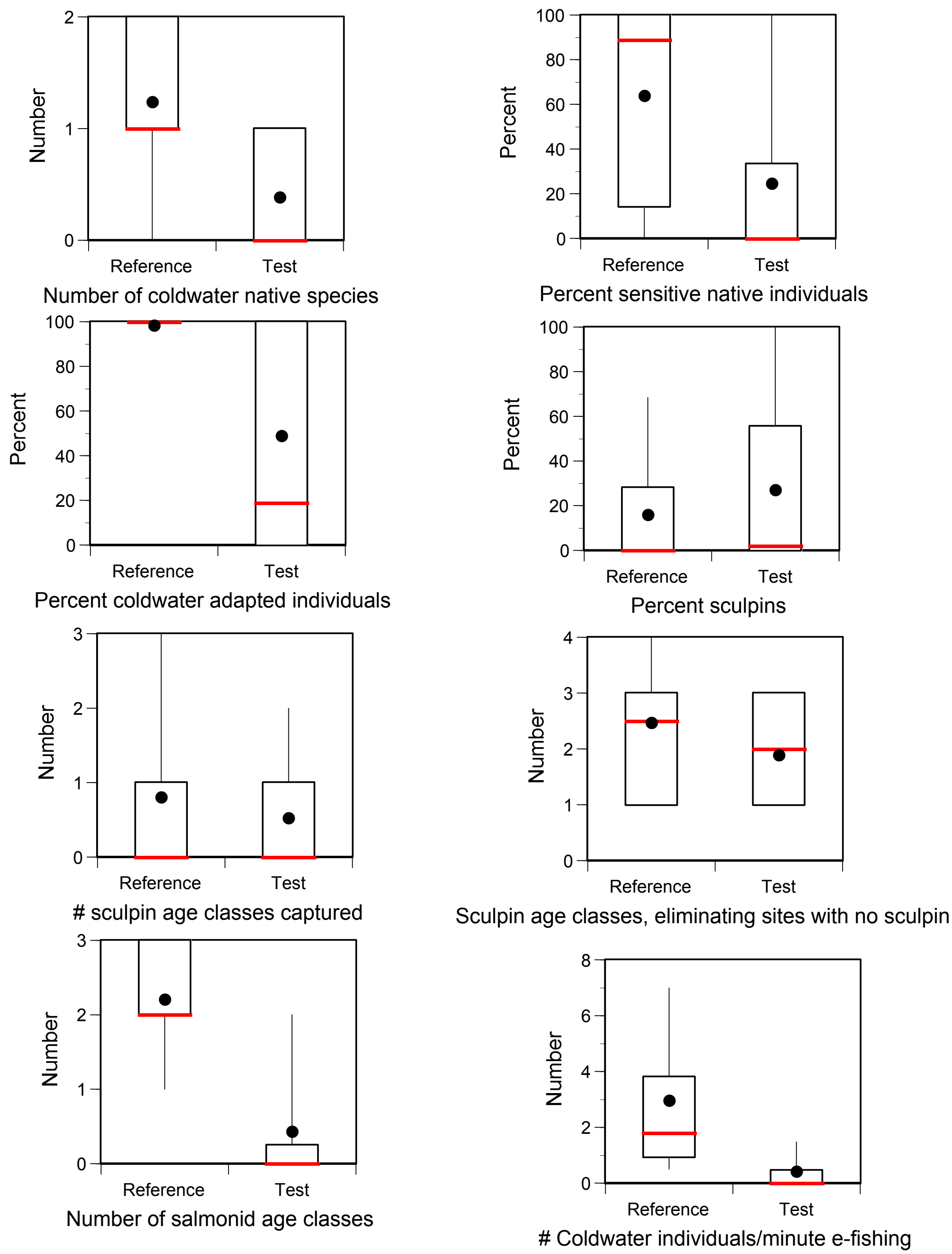


Figure 4-11. Small mountain streams: metric values for apparently least-impacted reference streams and apparently impacted test streams (Small streams = ≤2nd order).

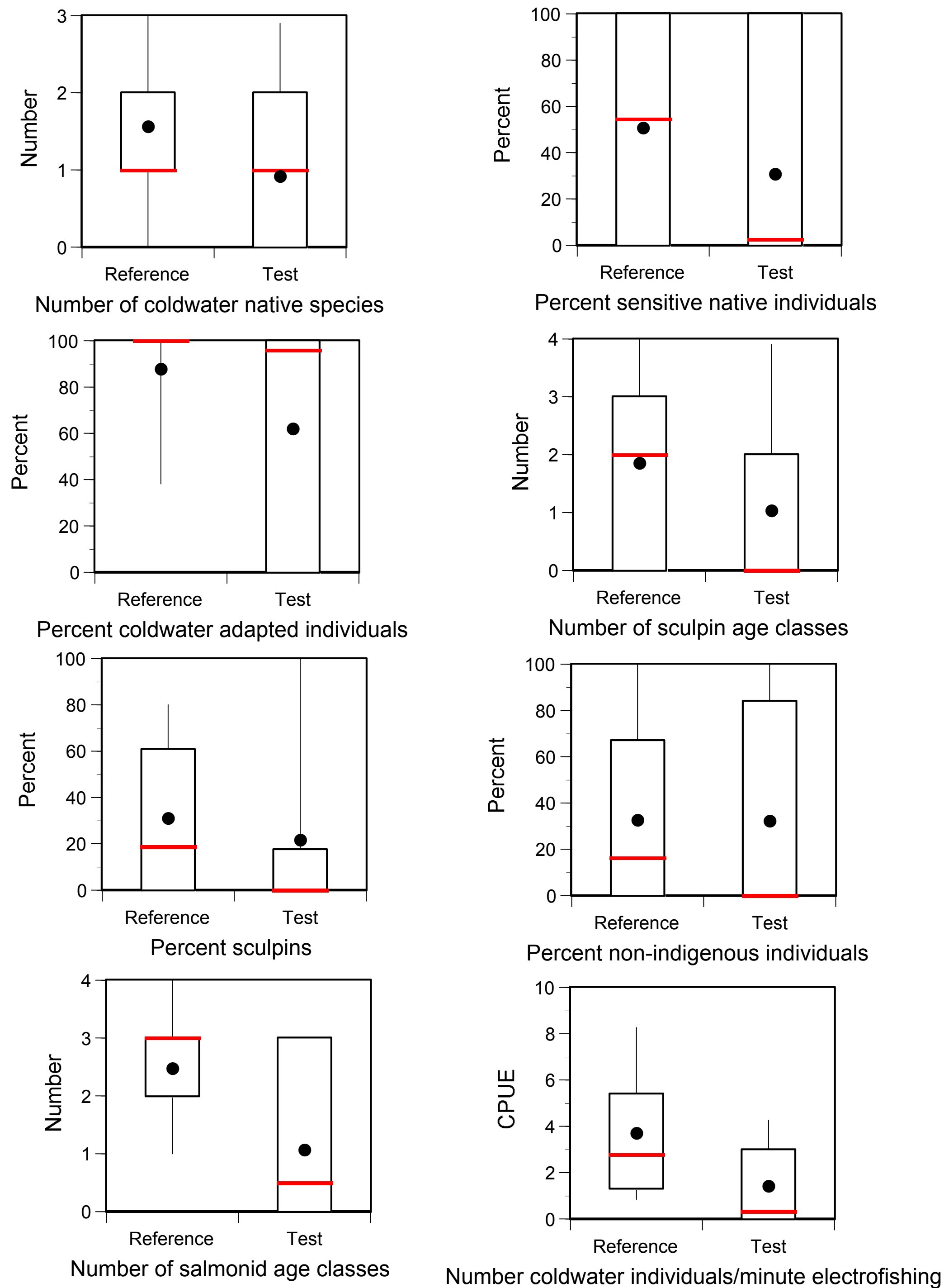


Figure 4-12. Intermediate sized mountain streams: metric values for apparently least-impacted reference streams and apparently impacted test streams (Intermediate size streams = >2nd order).

Percent sensitive native species – The representation of sensitive native species (e.g. native trout, salmon and char, shorthead sculpin) was variable, although sensitive natives were common in reference sites, and much less common in apparently impacted test sites (median percentages were 50 and 80 percent at small and large reference sites, versus 0 to 5 percent at small and large test sites respectively (Figures 4-11 and 4-12). Since they are considered sensitive to overall pollution, presence of even a small proportion of sensitive natives should indicate high quality water. Thus, the metric is scaled to increase steeply, and tapers off to 1.0 at high percentages (Figure 4-15). In other words, an assemblage made up of 40 percent sensitive natives is not assumed to reflect significantly lower water quality than an assemblage composed of 100 percent sensitive natives.

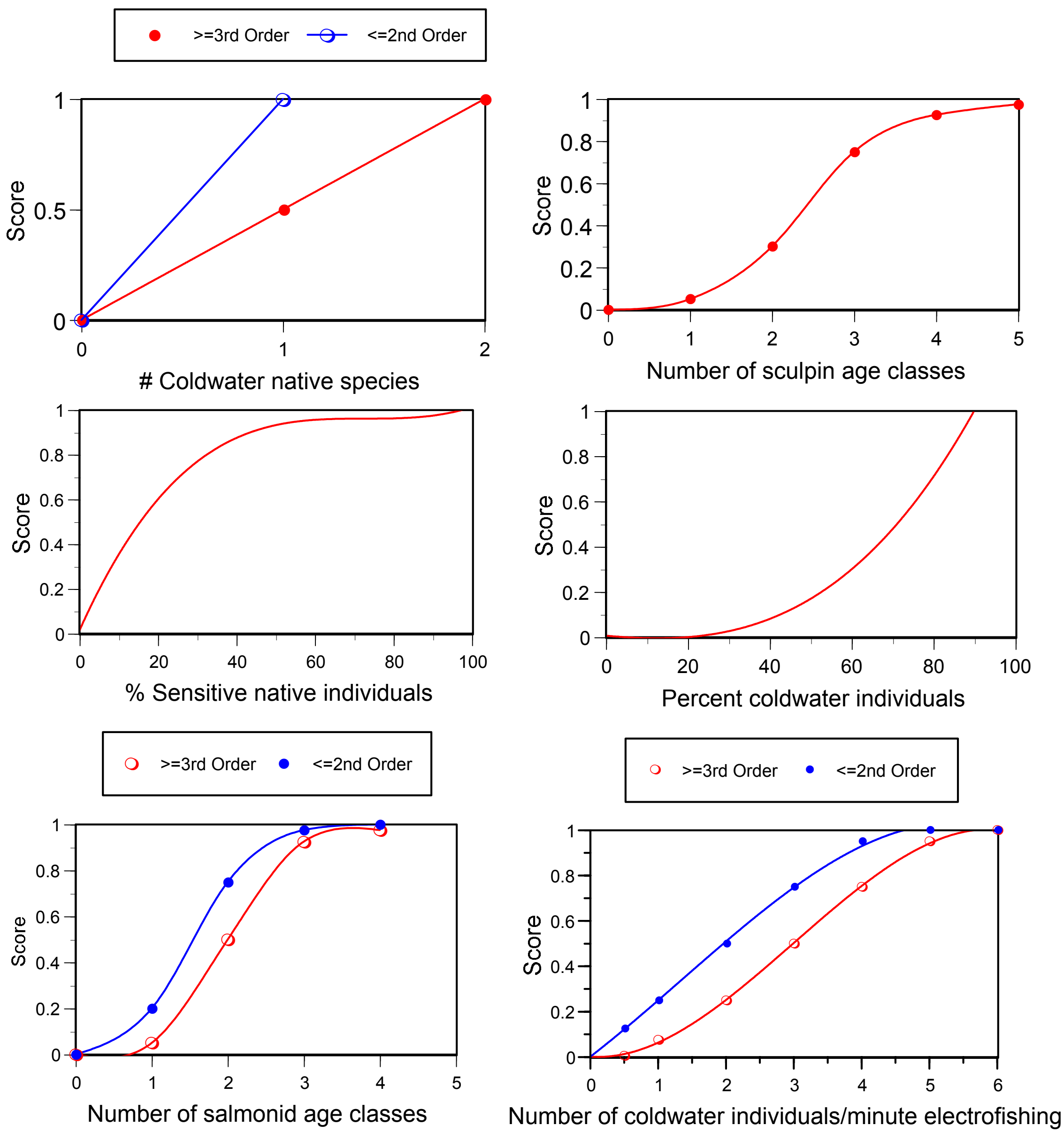


Figure 4-13. Scoring curves for metrics used in the mountain streams IBI.

Number of sculpin age classes – Multiple age classes of sculpin were encountered somewhat more often at reference, than test sites, particularly in the larger streams (Figures 4-11 and 4-12). The metric is scored on a "S" curve. It assumes that several age classes of sculpin would be present at reference sites, and if only one or two age classes are captured, that suggests that the missing age classes were not captured because they are rare or are missing. Shifts in age distribution are a common response to environmental stress on fish populations (Munnkittrick and Dixon 1989). Here, number of age classes captured is assumed to be a rough approximation of age distribution shifts. If environmental conditions reduce year-year survival and cause a shift to younger ages, or recruitment failures results in a shift to older ages, the old or young age classes will become more rare, and less likely to be captured. The metric therefore also assumes limited electrofishing efficiency, particularly in a single or 1st electrofishing pass. If one could enumerate every fish in a stream, if a species is present, more than a single age class will usually be present. The age class metric provides a rough indication of the relative abundance of different age classes. Since sculpin are sedentary benthic species that require cavities to nest in which are not highly embedded with fine sediment (Table 4-2), presence of multiple age classes is considered an indication of reasonably stable substrate without excessive sedimentation.

Interpretation of the usually ubiquitous sculpins in mountain streams is complicated since their absence could reflect *either* natural conditions or impairment. In high gradient streams (≥4%), salmonids may naturally be the only family of fish present (Figure 4-7). Therefore, in those streams, the absence of sculpin does not suggest anthoropogenic impairment; their absence is simply part of the natural variation. However, sculpin may be scarce or extirpated from disturbed streams, even though "sensitive" salmonids may still be present and even reasonably abundant. This pattern has been reported for metals (Carline et al. 1994, McCormick et al. 1994) and fine sediments (Mebane 2001). Since sculpin are commonly naturally absent from high gradient streams but usually abundant from forest streams with more moderate gradients, if for the higher gradient streams sculpin are not found, this is presumed to be natural and the sculpin age class metric is not included.

Number of selected salmonid age classes (trout, char, and salmon) – Multiple age classes of salmonids were encountered much more frequently at reference sties than test sites. Multiple (>2) age classes were encountered in the larger reference streams more often than the smaller streams (Figures 4-18, 11, 12). This pattern is presumably because of the greater space availability and space requirements for larger fish in the larger streams (Chapman 1966). Bjornn and Reiser (1991) report size segregation by salmonids in streams, with the older fish occupying deeper pools, forcing the smaller fish forced to shallow areas. Thus, the metric is scored so that the presence of 2+ age classes in small forest streams results in a high score, and 3+ ages in larger streams results in a high score (Figure 4-15). Mountain whitefish, which are excluded from this metric, were seldom encountered in the stream samples. As with sculpin, this metric could be considered a rough indication of an age distribution shift.

Relative abundance – The larger (3rd and 4th order) reference streams often had greater abundance of cold water individual fishes, than did the smaller streams (Figures 4-9, 11-12). The metric scoring is scaled accordingly, e.g. capturing three cold water fish/minute of fishing in a small stream would be scored as 0.7 points, and 0.5 on a larger stream (Figure 4-15).

Density or biomass, interpreted alone, can be a misleading indicator of habitat quality (van Horne 1983). However, in this case, relative abundance is used as one indicator of habitat or water quality, among other metrics. In addition, this measure relative abundance of cold water fishes usually discriminated well between reference sites and test sites (Figures 4-11 and 4-12).

The multimetric index for forest streams

The metrics were combined into a multimetric index on a scale from 0 to 100. The index appears to perform adequately, with most reference sites scoring higher than most test sites, with less than 25 percent of each group's scores overlapping (Figure 4-16).

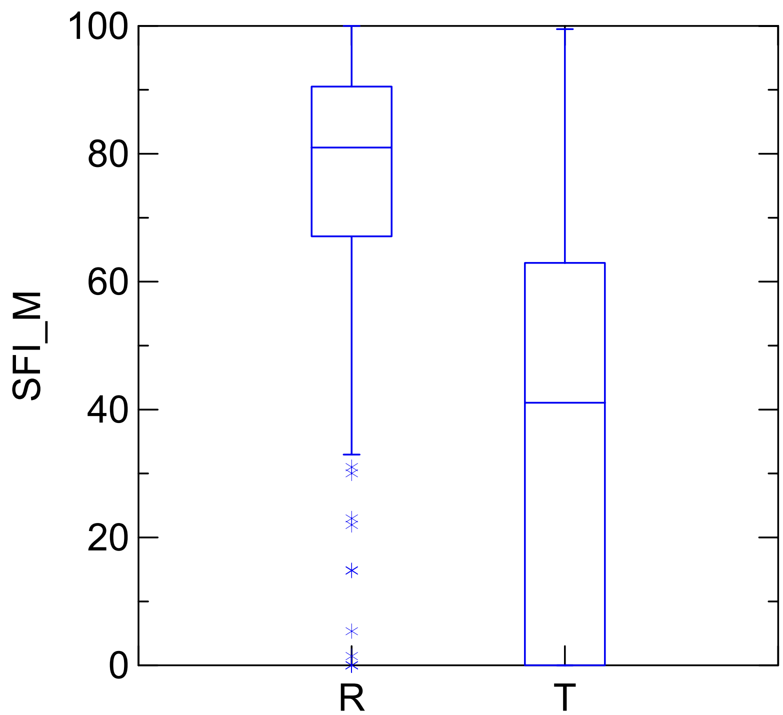


Figure 4-14. Ranges of forest SFI scores for reference (R) and test (T) sites.

Rangeland Streams

Few of the rangeland streams could be considered completely undisturbed by grazing or farming. Some of those with minimal human use were springs or were located in deep canyons and were probably not representative of non-canyon streams. Samples from springs were excluded from the index development. Maret (1997) showed that rangeland springs have a very different fauna from streams without discrete springs as their major source. Spring samples were identified by either being from known spring creeks, and from streams that (1) were named “Spring” and (2) examination of a topo map showed the sample was collected near the mapped spring source.

Rangeland streams were typically occupied by a mixed assemblage of cold water and cool water species (Table 4-4). Cyprinids were usually numerically dominant as number of individuals; as presence of species, there was often about an even mix of cold water and cool water species present. The metrics tested (and subsequent index) for these streams were similar to the original IBI concept and ecological structure, based on taxonomic richness, habitat, and trophic guild composition (Figure 4-15, Table 4-1). The median or mean values of several metrics responded as expected for reference or disturbed streams. However, most candidate metrics had considerable overlap between values for reference and apparently impaired test sites.

Few reference rangeland streams were believed to be undisturbed according to Table 4-3 criteria. Although many streams still have natural features, a century of grazing and water development may have fundamentally altered fluvial or potential vegetation features from their historical reference condition, resulting in low ecological integrity (Quigley et al. 1996). Thus, the possible loss of the historic reference condition may lessen the distinction between “reference” sites (i.e. best of what’s left) and “test” sites may lead to metric responses being dampened. Because of this potential, candidate metrics were included in the index development, based upon IBI concepts, even though two (number of native species and percent non-indigenous individuals) were not fully empirically distinguished. Curiously, in earlier preliminary analysis, reference sites tended to have higher native species counts than apparently impaired sites. However, after adding a large number of sites from IDFG surveys of the Owhyee Highlands streams, these differences vanished. Even though these results are ambiguous, the desirability of conserving native species suggests that measuring native species could be useful, and the metric is currently included in the multimetric index.

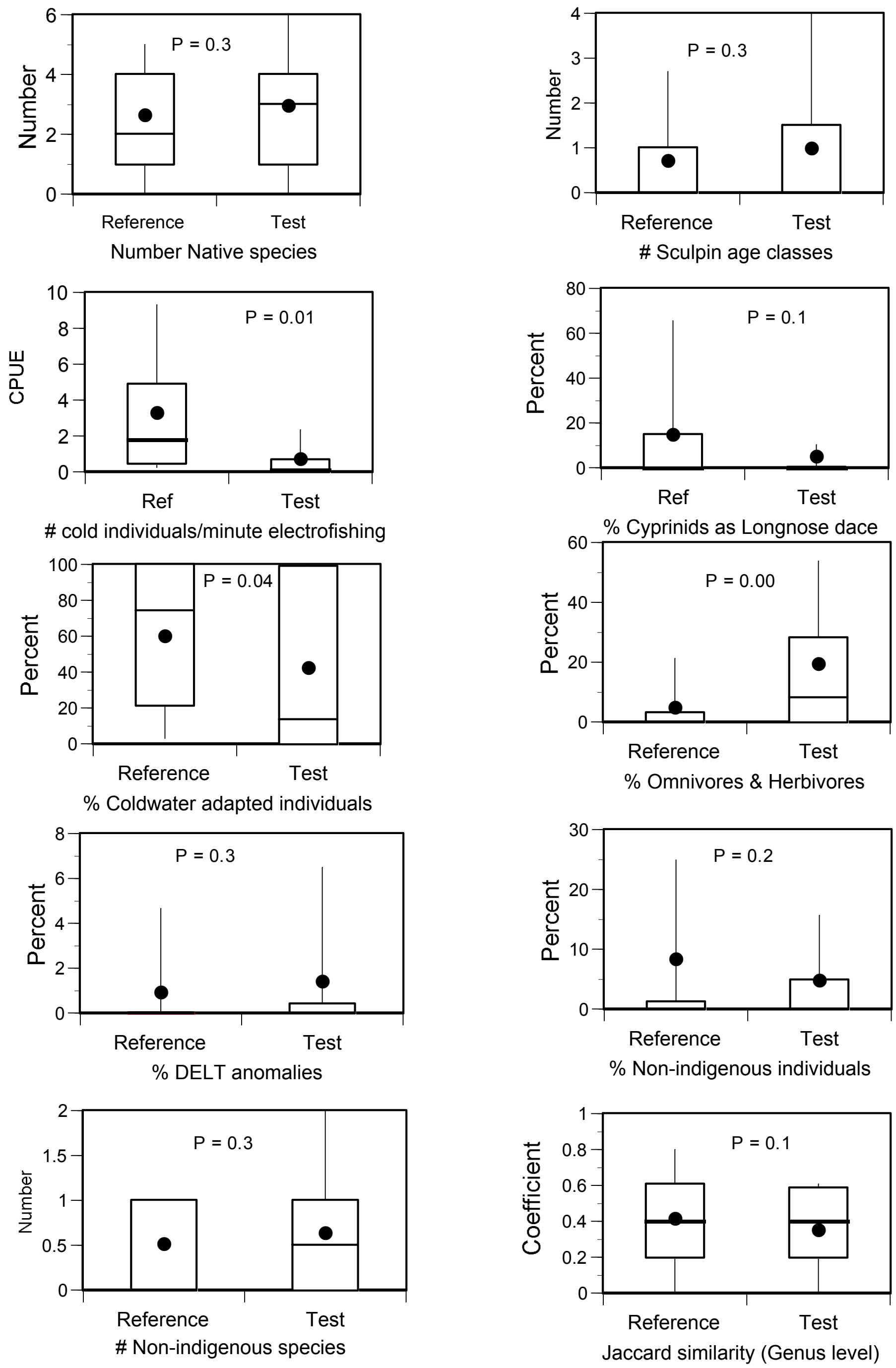


Figure 4-15. Candidate metrics ranges, for minimally and apparently impacted sites rangeland (basin) sites (n=51 and 32 for reference and test sites respectively).

Metric evaluation and scoring for rangeland streams

Six metrics were included in the rangeland stream fish index ("SFI-Range"):

1) Percent cold water individuals,

2) Of the cyprinids, percent that were longnose dace,

3) Percent omnivores and herbivores

4) Percent DELT anomalies

5) Jaccard community similarity coefficient, and

6) Number of cold water individuals captured per minute of electrofishing.

Rationale for the metrics selection and scoring follow.

Percent cold water individuals – This metric showed strong differences between reference sites, with a median of 70 percent cold water individuals; and test sites, with a median of 10 percent cold water individuals (Figure 4-15). The metric is scored linearly between 0 to 30 percent cold water individuals, reflecting the concept that rangeland streams are expected to naturally have a mix of cold water and cool water adapted species (Figure 4-6). Thus, a stream with 100 percent cold water species would not necessarily have higher biological integrity. Instead, it probably would have been misclassified.

Of the cyprinids, percent that are longnose dace – Native cyprinids (dace, shiners, chub, pikeminnows) were ubiquitous in rangeland streams, and were nearly always collected in perennial rangeland streams, both from disturbed and reference streams. Cyprinids occurred in about 90 percent of reference rangeland streams (Table 4-4). Among the native cyprinids, on the average longnose dace were found in higher percentages in reference streams than in disturbed streams. Longnose dace were seldom the dominant cyprinid (Figure 4-15). Longnose dace distribution may be limited by excessive sedimentation or flow reductions (Propst 1982). This metric is scored linearly over the range of expected percentage of occurrence in reference streams (0 to 15 percent of the cyprinids). If cyprinids are present, and no longnose dace occur, a score of zero results; if longnose dace make up 15 percent or greater of the cyprinids, a maximum score is assigned (Figure 4-16).

Percent Omnivores and Herbivores – Omnivores and herbivores comprised less than 10 percent of the sample in most rangeland streams, and they were more abundant in apparently impacted test streams than reference streams (Figure D-5, Appendix D). The metric is scored with a "ski jump" shaped curve which is flat on the top, steepens, and then is flat at the bottom to follow the expected patterns in rangeland streams (Figure 4-15). Omnivores and herbivores are expected to be present in rangeland streams but at low percentages; thus the flat top of the curve gives a maximum score up unto a threshold of 10 percent. From 10 to 40 percent, the scoring curve steeply drops, then flattens out to reflect that streams with greater than 40 percent omnivores and herbivores are very different from reference and get a low score. Conceptually, this “ski jump” curve is similar to having a straight line dropping from 10 to 40 percent, or using the “5,3,1” scoring bin approach, where the reference condition (less than 10 percent) is given a score of 5, the departure from reference zone (10 to 40 percent) is given a score of 3, and the significant departure from the reference condition (greater than 40 percent) is given a score of 1. The “ski jump” curve has the advantage of providing continuous scores rather than having a large, and probably not biologically meaningful, score jump between 5 and 3 for values of 9.5 percent and 10 percent.

Percent Deformities, Eroded Fins, Lesions, and Tumors (DELT anomalies) – DELT anomalies were rare in all sites, but were a little less rare in test rather than impacted sites (Figure 4-15). The metric is scored as a steeply declining exponential curve, to reflect the expectation that in reference sites, less than 1 percent of the fish samples are expected to show anomalies (Figure 4-5). The inclusion of the DELT anomaly metric is also supported by other studies from similar ecoregions. Munn and Gruber (1997), working in the Columbia Basin ecoregion of Washington and Idaho, found increased detections of organochlorine pesticide residues in fish in watersheds with high percentages of agriculture. Examination of their raw fish assemblage data[10] also showed higher prevalences of DELT anomalies than in non-agricultural watersheds. However, no quantitative dose-response relationship between the residue concentrations or frequency of detection was apparent. Fish samples from rivers in Idaho with high percentages of agriculture upstream of the sample location also had higher occurrences of DELT anomalies (Mebane 2000).

Jaccard community similarity coefficient (JC) – This metric is commonly used in ecological studies to measure the degree of similarity in taxanomic composition between two stations. In this case, we are interested in the comparison to the average taxa composition for rangeland reference streams, not individual stations. Chandler et al. (1993) and Stevenson and Bahls (1999) recommend using average community composition of a group of reference stations to calculated compositional similarity indices.

To calculate the coefficient, all taxa counts were transformed to binary taxa presence or absence data . The five most frequently occurring genera (Table 4-4) at all the reference sites for the rangeland ecoregions were considered the taxa likely to be present at streams with species composition similar to that of reference streams. In the Jaccard coefficient (Table 4-2), these five most commonly occurring genera were treated as the taxa present at the reference “station,” and then using, the taxa composition of each comparison site (those sites described as “test” or “other”) Jaccard coefficients were calculated. Similarity coefficients ranged from 0.0 to 0.7, with most values ranging between 0.1 and 0.5 (Figure 4-5). The metric was scored linearly with JC of 0 to 0.6 equating to a score of 0 to 1. The value of 0.6 was selected as the maximum score based upon the following factors:

- The range of the data
- Community samples may be noisy
- Replicate fish assemblage samples typically have a similarity range between 0.6 and 1.0 (Maret 1997, Gauch 1982)

[10] Provided by M. Munn, U.S. Geological Survey, Tacoma, WA

Since the comparison value is from a group of reference stations rather than a single reference station, the likelihood of a single comparison station having all the reference species present is low, so the lower bound of the range of reported replicate similarities would more likely actually occur.

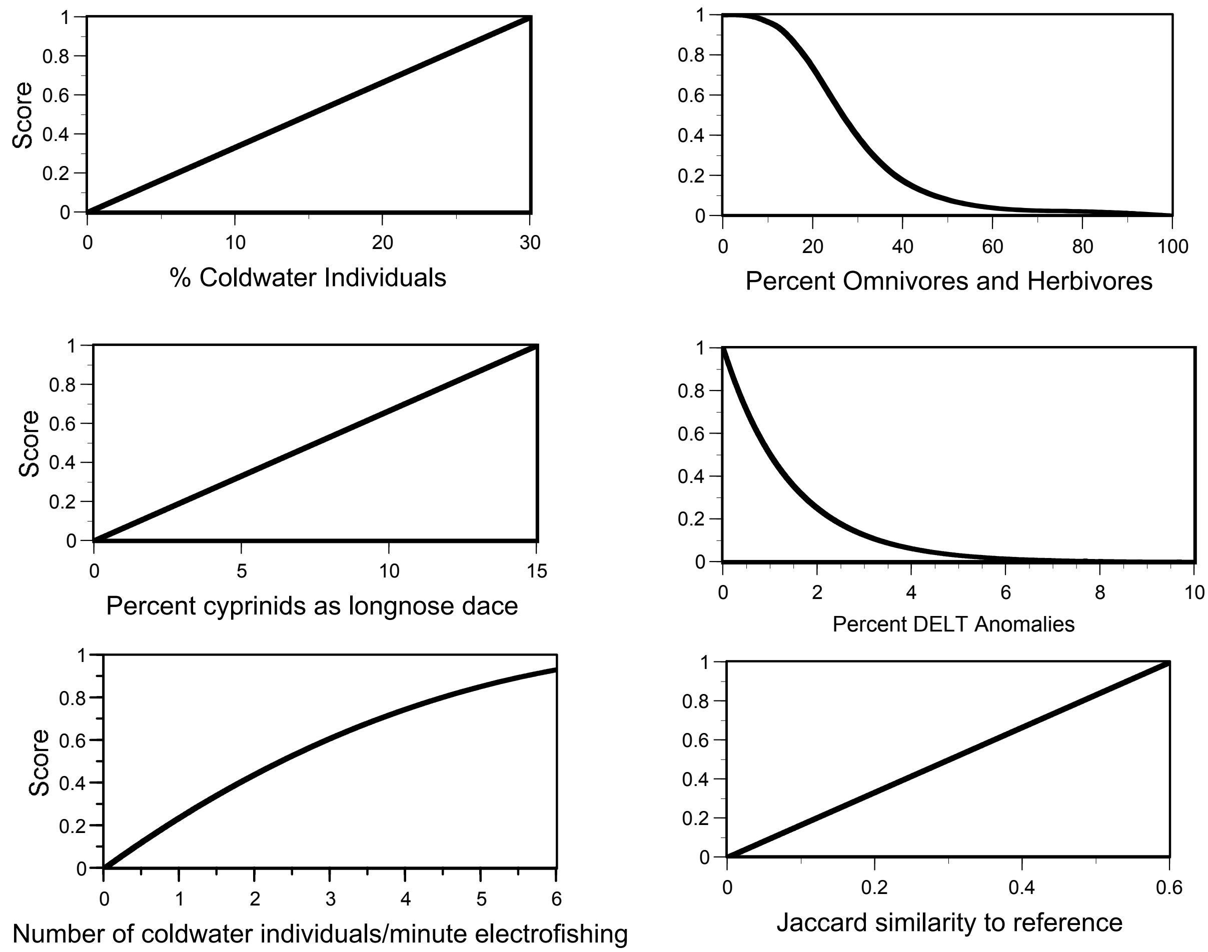


Figure 4-16. Scoring curves used in the rangeland stream fish index.

The multimetric index for rangeland streams

As with the forest streams, to combine the individual metrics into a multimetric index, the unitless metric scores were added together and adjusted to have a maximum possible score of 100. Metric scores were weighted in the index according to how well they discriminated between reference and impaired sites. Metrics that discriminated relatively weakly ($P < 0.1$) were multiplied by 0.5 before being added together into the index. This was done to reduce the influence of these metrics on the overall score, compared to the more strongly discriminating metrics[11].

[11] No weighting was necessary in the forest index because all selected metrics discriminated between reference and test sites at $P < 0.05$.

When combined in this manner, and summarized with box plots, the index performed reasonably well (Figure 4-17). Scores for reference sites ranged from 50 to 100, with a median score of about 82. Scores for test sites ranged from 0 to 90 with a median score of about 55. If we consider the body of the data to be the inside of the box plot boxes (i.e. the interquartile range from the 25th to 75th percentile of scores), then we note that there is some overlap of the boxes, indicating that the index's efficiency to distinguish apparently impaired and reference sites is somewhat limited.

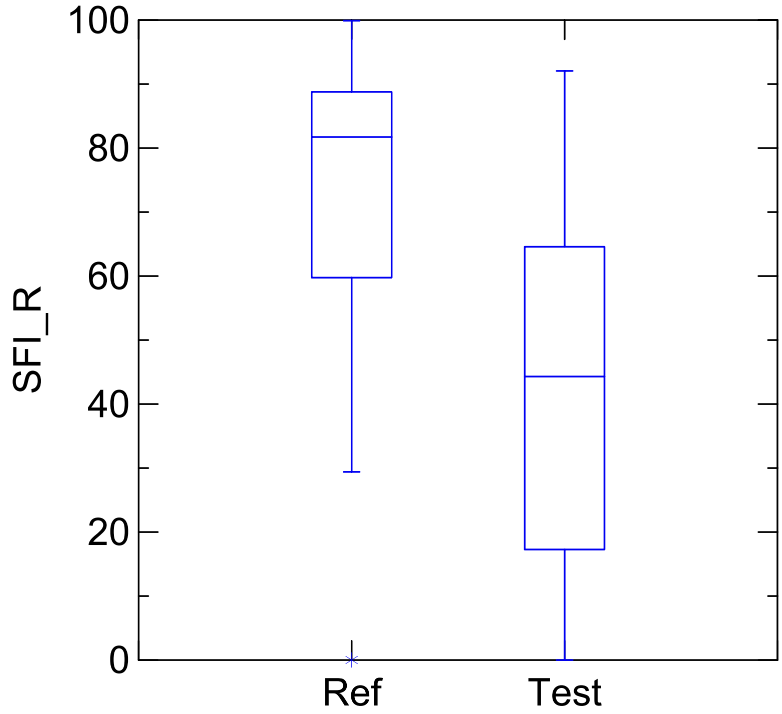


Figure 4-17. Ranges of rangeland fish IBI scores for reference (R) and test (T) sites.

Classification of Stream Types – Revisited

Discriminant analysis of stream classes using index metrics

The previous results (Figures 4-5 to 4-16) used simple scatter and box plots to examine the data and evaluate fish assemblage variables to see if they would serve as meaningful metrics, i.e. variables that responded to environmental degradation. Whereas initially, the stream classifications were evaluated with discriminant analysis to see if reference sites differed in terms of fish species composition (Figure 4-4), now we are considering whether using the values of 11 metrics from both indexes, the streams separate into the four pre-defined groups: forest streams (reference and test), and rangeland streams (reference and test). As with the groupings by species composition in Figure 4-4, the stream class groupings by using metrics have both shared and distinct characteristics (Figure 4-17). Forest reference streams are tightly clustered in the center of the plot, but overlapping clusters show that some forest test streams and some rangeland reference streams share characteristics of the forest reference streams, and with each other. Both the rangeland reference streams and the forest test streams are more variable than the forest reference streams. The rangeland test streams are most variable of all, but also most distinct from the other groups.

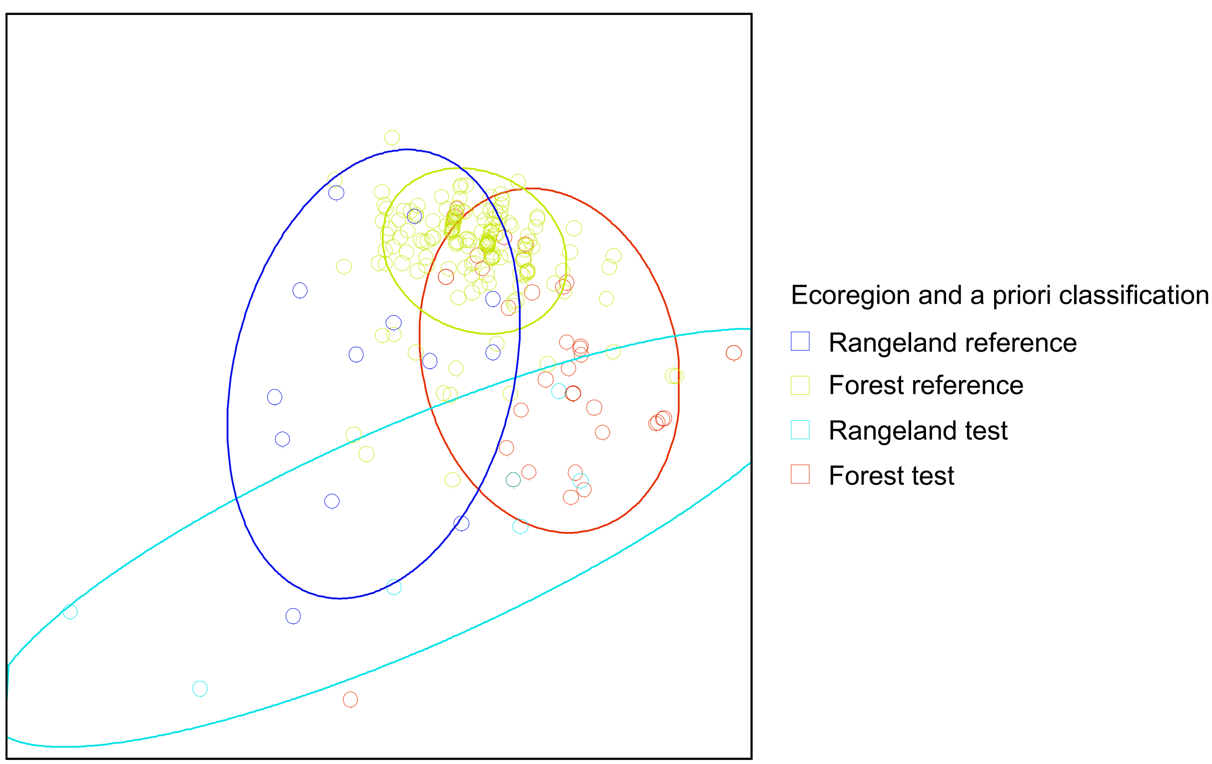


Figure 4-18. Canonical correlations of all fish metrics from both indexes grouped by reference and test classifications and by forest and rangeland ecoregion groups.

Correlations between watershed disturbance, habitat, and fish metrics: methods

A companion project to this one is to evaluate stream habitat measures and to revise a stream habitat index provides information relevant to the stream fish indexes (Fore and Bollman 2000). Idaho currently has a habitat index for streams consisting of 11 quantitative and qualitative channel morphology, riparian, and substrate features (DEQ 1996; described in Mebane 2001). Fore and Bollman (2000) tested this index, and about 36 individual stream habitat measures with measures of disturbance at the site and watershed scale, and with biological conditions at the sites. They then developed a new habitat index using the best component metrics. The habitat measures were largely evaluated by testing for significant correlations between land cover and land use measures, and macroinvertebrate or fish metrics. The fish metrics they used were based on an early version of this report, and so are similar to those used here in the forest/montane SFI. The rangeland SFI could not be compared to Fore's and Bollman's results, although some of the metrics could be compared.

Both Fore and Bollman's work and this effort included evaluations of the responses of fish metrics to human disturbances, but with different approaches and using data from different sites. In this report, a qualitative checklist of disturbance factors was used with interviews with local biologists to identify apparently reference or impacted streams (Table 4-3). In contrast, Fore and Bollman supposed nothing *a priori,* and based their evaluation primarily on correlations between quantitative stream and watershed measures. They were provided matched habitat and fish data from DEQ surveys for 140 sites based on 2 factors:

1. Sites were targeted from the typic parts of their ecoregions as mapped by Omernik and Gallant (1986), in order to exclude transitional sites near borders of ecoregions.

2. Sites located near the most downstream point of the watershed, so that they could relate land cover data (which was calculated at the watershed level) to stream site condition data.

Further, the data Fore and Bollman used are from different sites than in this report, except for a few sites, which by chance overlapped.[12] Because of the different approaches, and different data used, the results can be compared without circularity. Also, using their data set and the forest/rangeland stream classification scheme, I tested the stream fish index metrics for correlations with their new stream habitat index, and with land use. Land use was evaluated as percentage of the watersheds in agricultural use (cultivated crops, not including open range grazing) because of previous regional research showing significant biological relationships to broadscale agricultural use but less so to other common land uses (e.g. rangeland, forestry, urbanization) (Maret 1997, Schomberg et al. 1998).

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[12] As of the end of the 1999 field season, fish data from over 1000 stream sites had been collected as part of the DEQ BURP program. At the time this project started, the data were not readily available from the BURP relational database, and so raw data from over 400 of those sites were entered into a spreadsheet and metrics and index scores were calculated for them. Fore and Bollman's data including the fish metrics were provided from the BURP database. Twelve sites common to both projects were checked to see if the metric values matched. They did. The only exceptions were age classes which often differed by 1. This difference is attributed to the difference between database calculating the estimated age classes using size bins from Appendix 3, and here, where they were estimated by size bins and by judging clusters of sizes.

Correlations between watershed disturbance, habitat, and fish metrics: results

The percentages of the watersheds in agricultural use were significantly correlated with several fish metrics when all sites were pooled (Table 4-5). No correlations were significant when evaluated separately for the forest and rangeland sites, or when evaluated separately for only the Northern Basin and Range (NBR) ecoregion (which by chance is where most of the agricultural watersheds fell). However, the correlations for the rangeland or NBR sites were in the expected direction, and with sample sizes of only 16 and 12, the lack of significant correlations is not surprising. There is very little agricultural use in the forest/mountain streams' watersheds.

Correlations between fish metrics and habitat were often present. Generally, forest streams showed stronger correlations with the habitat index, than did the rangeland streams (Table 4-5). Component habitat measures tested against fish measures by Fore and Bollman (2000) varied greatly by ecoregion (Table 4-5, "Individual Habitat Measures" column). For example, the number of salmonid age classes was often significantly correlated with either the habitat index or component measures, but with only one individual measure in the sites from the Northern and Middle Rockies. This latter result was puzzling since salmonid age classes performed well here in forest streams (Figures 4-11 and 4-12). Using the same data set as Fore and Bollman, I tested correlations between several of the better performing habitat measures and salmonid age classes for forest streams. Rather than one, there were six significant ($p<0.05$) correlations with salmonid age classes: large woody debris, percent substrate less than 2mm, Wolman substrate size classes, channel shape, pool:riffle ratio, and percent canopy cover. Since these differences were even more unexpected, I re-ran the correlations using the same ecoregional stratifications as Fore and Bollman. The resulting Spearman coefficient values were identical to those reported by Fore and Bollman. The remaining difference between our tests is that they stratified streams using ecoregions, and I stratified streams using a combination of ecoregions and elevation. The differences remain puzzling but intriguing.

While statistical significance does not always indicate biological significance and the converse, the above patterns suggest some metrics are often significant in a variety of circumstances.

Table 4-5. Pearson correlations for fish metrics, watershed disturbance, and Habitat index ($p<0.1$ = +, $p<0.05$= ++); and numbers of significant associations between the metrics and with discrete habitat measures .

Metric	Land Use	Habitat Index		# of Significant correlations with Individual Habitat Measures (36 tested)		
Ecoregion (Sample size)	Pooled (84)	Forest (89)	Rangeland (45)	NR/MR (varies)	SRB (varies)	NBR (varies)
Native taxa richness		++		6	5	1
% Non-indigenous individuals				5	4	1
% Omnivores and Herbivores		+		1	1	1
# Cold Native species	++	++		3	10	1
# Sculpin age classes	+	++	++	4	6	1
% Sensitive native individuals	+			1	10	5
% Cold individuals	++	+		2	7	6
# Salmonid age classes	++	++	++	1	9	17
CPUE of cold water individuals		++		2	10	3

Table sources: Land use and Habitat Index calculated using same data set as Fore and Bollman (2000). Individual habitat measures were compiled from Fore and Bollman 2000, appendices 1-3.

Table notes: NBR- Northern Basin and Range; SRB-Snake River Basin; NR/MR – Northern and Middle Rockies; see figure 1 for locations.

Individual habitat measures- Fore and Bollman used Spearman rank order coefficients, with $p<0.05$ = significant associations. Since the Systat® statistics program used here did not support probability calculations for Spearman correlations, Pearson correlations were used. The r values between the Spearman and Pearson coefficients were very similar for all tests, suggesting the probabilities likely are similar too.

Comparison between multimetric fish and habitat indexes

Comparing the forest fish index, rather than individual metrics, to the habitat index, shows that sites with high habitat scores tended to be accompanied by fish index scores that were higher and less variable than sites with low habitat scores (Figure 4-18). Splitting the habitat scores at their medians, and plotting the above and below the median fish scores, shows an association between the habitat and fish indexes. Sites with high habitat index scores tended to be accompanied by fish index scores that were higher and less variable; fish index scores were more variable and somewhat lower at sites with low habitat index scores. Habitat scores were considered “high” or low if they were above the median value.

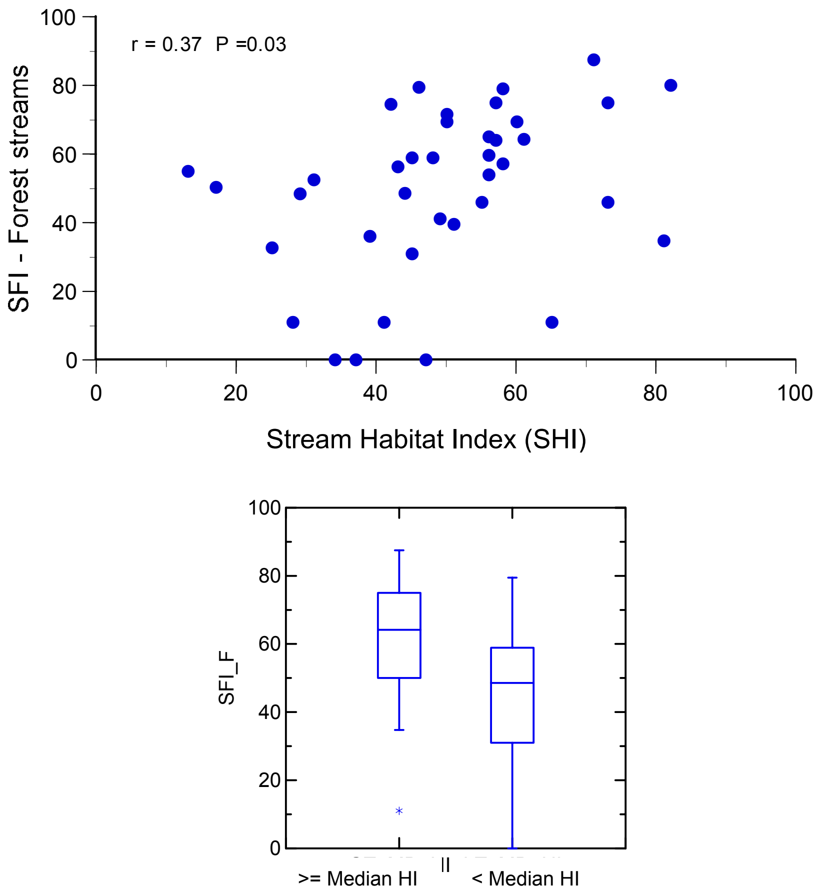

Figure 4-19. Habitat and fish index relationships for forest streams.

These results show that habitat and fish relationships are often apparent, but are inconsistent. Explanations for this are elusive – do the inconsistencies result our inability to define and measure the correct fish and habitat attributes? Are fish-habitat relationships highly variable? Is it measurement error, problems of scale, or more than one of the above? Assessment scale seems likely to have some bearing on the noise in the above relationships. Much of the above data is at a site scale (100m reach), and could be influenced by access to refugia or disturbance at a stream scale. As Dunham and Vinyard (1997) note "ignoring stream [scale] effects can lead to erroneous conclusions about site level habitat variables."

Stream fish indexes and sediment comparisons

Excessive sedimentation is a significant cause of water quality impairment in North America. In the United States, 34 percent and in Idaho, 93 percent of waters listed as impaired are attributed to excessive sedimentation (EPA 1994, 1995b). The particle size of deposited stream bed sediments affects the flow resistance in the channel, the stability of the bed, and the amount of available aquatic habitat types. These in turn have significant effects on macroinvertebrate and fish communities (Minshall 1984; Waters 1995).

Thus the substrate is a fundamental part of the stream environment, and siltation is a ubiquitous concern. However, there is no consensus method for characterizing sediment in streams for biological surveys. Over 10 methods for evaluating sediment in streams are in use in North America, ranging in complexity from visual estimates to liquid nitrogen freeze cores (McDonald et al. 1991). Our program uses Wolman pebble counts to characterize substrate particle sizes. Pebble counts of stream channel transects were developed as a method of characterizing the particle size distribution of the stream bed in order to calculate stream hydrology features: flow resistance, channel capacity, and streambed stability (Wolman 1954, Kondolf 1997). Recently, pebble counts have been recommended as an efficient, and repeatable means for evaluating the suitability of stream substrates for aquatic life (Fitzpatrick et al. 1998; Conquest et al. 1994; Bauer and Burton 1993; McDonald et al. 1991). However, few tests have been reported between surface sediment sizes and results have been inconclusive. In Mebane (2001), I found that high percentages of fine grained surface sediments interrupted salmonid and sculpin life cycles. Sculpins consistently avoided areas with abundant fine sediments in the stream channel or deposited on the banks — the more motile salmonids only responded to fine sediments in the stream channel. In contrast, Fore and Bollman (2000) reported that correlations salmonid age classes and surface fine sediments were variable, highly significant in the Snake River and Northern Basin and Range ecoregions but not in the Northern Rockies ecoregions.

Here, I grouped forested sites by whether their index scores were in the range of reference sites, using the 25th percentile of scores from reference sites as the threshold for whether the sites were similar to reference conditions. Then the ranges of fine sediment, measured either across the submerged portion of the stream channel (instream) , or measured across the bankfull width (instream plus bank deposits) were compared from sites with scores similar to reference to those with lower scores (Figure 4-19 top). In both cases, sites with higher SFI scores tended to have lower percentages of fine grained (less than 6mm) sediments. Similar comparisons with the Fore and Bollman (2000) data set, were not statistically significant although the plot of bankfull sediment less than 6mm versus SFI had the same direction of differences. Unfortunately, instream fine sediments less than 6mm were missing from their data set, so like comparisons could not be made. The insignificant results for SFI vs. instream fines less than 2mm might be related to the practical difficulty of measuring very small particles in pebble counts, small sample size, or the lack of biological significance. The latter seems unlikely since many workers have reported on the relatioship between fine sediment and salmonid fish populations, although there is less agreement on what the best threshold for “fines” are for this purpose, ranging from less than 10mm to less than 0.85 mm (Chapman and McCleod 1987, Chapman 1988, Waters 1995).

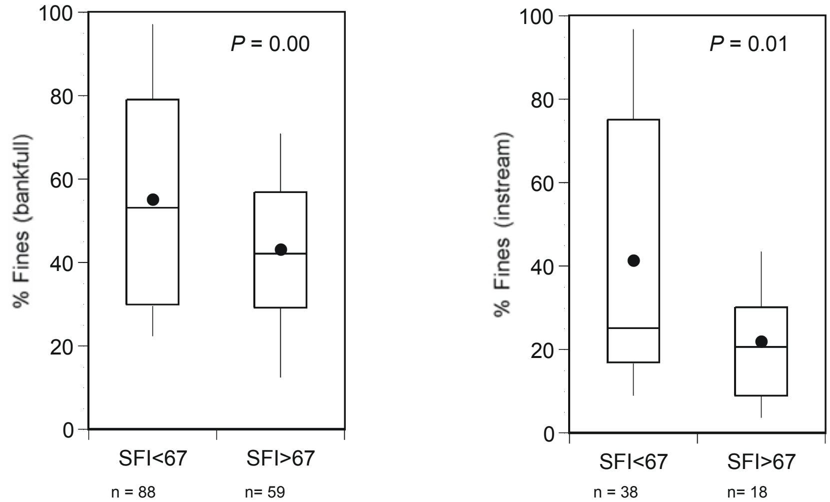

Figure 4-20 (top)

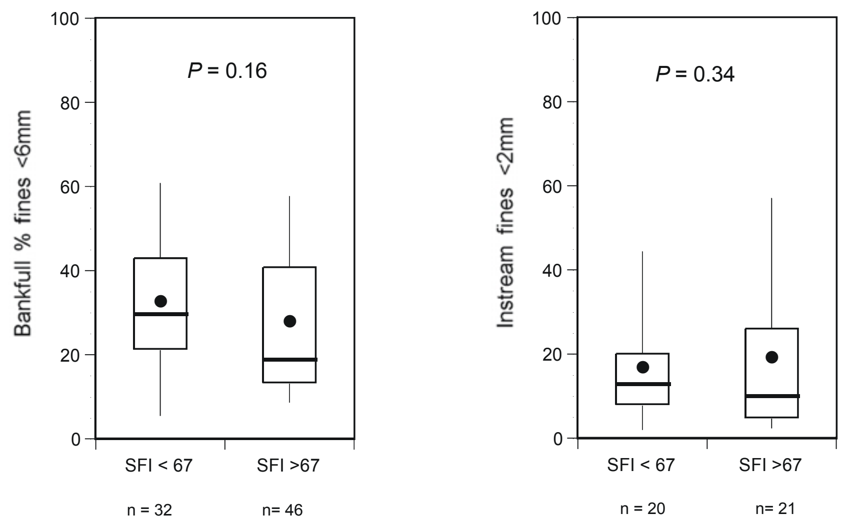

Figure 4-20 (bottom)

Figure 4-20. Top: Percentages of channel widths covered by fine-grained (<6mm) sediment occurring at sites with SFI scores <67> (forest streams).
For these tests, sites with SFI score >67 are considered similar to reference condition, scores <67 are considered potentially impaired (67 was the 25th percentile SFI score for reference sites). "Bankfull" includes bed materials located both in the submerged channel and on the banks; "instream" includes bed materials from the submerged portion of the channel only.
P values indicate the probability that the means are equal using t-test.
Bottom: Similar comparison using the Fore and Bollman (2000) data set. Percentages of bankfull channel widths covered by <6mm sediment and instream width covered by <2mm surface sediments. Data for instream fines <6mm were not readily available

Native Amphibians: A Supplemental, Qualitative Metric

Had amphibians been consistently captured and identified in the data, quantitative amphibian-related metrics would have been evaluated. They were not. However, inclusion of amphibians into an IBI has several desirable features. First, the presence of native amphibians may be a qualitative indication of functional riparian and instream habitat. Further, in cold water streams with low fish species diversity, broadening the fish assemblage to a vertebrate assemblage of fish and amphibians would increase the potential species diversity somewhat. Lastly, amphibians have been in general decline in recent years (need cite), and state-wide systematic observations of their occurrences could provide useful information on their status.

The presence of native amphibians is scored as bonus points in either the rangeland or the forest stream index. Hopefully, by including native amphibian presence in the index, that will provide useful information on riparian and stream habitat condition, and perhaps more importantly, encourage future survey crews to better seek out (and key out) amphibians in riparian and stream habitats. Note that this qualitative metric is a “presence” metric, not a “presence or absence” metric. Because amphibian distributions may be naturally patchy, because some survey crews did not attempt to capture amphibians, and because many amphibians are riparian, rather than aquatic species and thus may not be captured in routine electrofishing, it would not be right for the absence of native amphibians to cause a lower index score. That is, supplemental points are added to the final index scores, absence of amphibians is not considered a negative factor, and their contribution to index scores were not considered in any index testing or establishing biological impairment thresholds.

In particular, one native amphibian, *Ascaphus truei*, the tailed frog, is highly specialized for life in clear, cold mountain streams. Tailed frog populations have declined in the Pacific Northwest primary because of timber harvesting (Bull and Carter 1996, Waters 1995). Declines in tailed frog populations are probably because of higher water temperatures, increased siltation, or greater bed movement. They are the only amphibian species of the Pacific Northwest known to spawn in fast moving water, attaching their eggs to the bottoms of rocks. Other amphibians usually spawn in lentic, or semi-lentic, slow moving water. Three age classes of tailed frog tadpoles will occur in occupied streams, they do not usually metamorphose until the end of year-four (Nussbaum et al 1983). In a study of 80 streams in the Blue Mountains ecoregion of northeast Oregon, Bull and Carter (1996) found that the percentage of cobble and boulders, proportion of the stream with at least a 30m buffer on both sides from roads or timber harvest, and stream gradient were strong predictors of tailed frog presence or absence from their study area. They also found clear trends of decreasing adult frogs with increasing timber harvest. No frogs were found in 30 percent of streams which had received a heavy amount of timber harvest, whereas all streams that had received a low or moderate amount of timber harvest contained frogs. Bull and Carter conclude that that timber harvest would unlikely significantly influence populations if a no-cut buffer was retained and the integrity of the stream structure was retained. The natural distribution of the tailed frog is another reason for treating its presence as a positive/neutral factor, rather than a positive/negative metric. According to Nussbaum et al. (1993), tailed frogs are widely (but unevenly) distributed across the areas shown in Figure 1 as the Northern Rockies and Blue Mountain ecoregions, but are not known to occur in other ecoregions of the state.

Because of these unique characteristics that specifically relate to stream quality, presence of tailed frogs is weighted more heavily than other native amphibians (10 points added to the base stream fish index of 100 points for tailed frogs, 5 points for other native amphibians).

DISCUSSION

The purpose of this analysis was to examine fish data from small streams in Idaho (primarily 2nd to 4th order) and, if feasible, construct a multimetric index reflecting the biological condition of the assemblage. The results so far indicate that multimetric stream fish indexes are feasible, and may be a useful interpretive tool for stream bioassessment. Considerations for the use of the indexes in water resource management and possible improvements follow.

Considerations for Setting Impairment Thresholds

EPA has defined a water body as being biologically impaired if the attributes of one or more major biological groups (assemblages) has been modified significantly beyond the natural range of the reference condition (EPA 1995). This definition provides a rational, general starting place. However, at the state or regional scale, a more explicit definition is needed. For example, with a large data set, the range of values of any measure will be large. In the case of both the rangeland and forest stream fish indexes, the range of scores at reference sites was from 0 to 100. Rather than these absolute ranges, statistical distributions of the natural range of reference conditions should be considered. Still, statistical definitions do not relieve the assessor from making a judgement of how significant a "significant modification" is. Perhaps conservation biologists may give quantitative advice on the necessary proportion of streams with fish assemblages in their natural reference condition to maintain viable metapopulations. More often, an impairment threshold will likely need to be made based upon social or policy judgements.

Approaches vary with different situations, investigators, and decision makers. Kilgour et al. (1998) argued that the 5th percentile of the normal distribution provided an objective, and natural definition that was supported by a nearly universal custom of selecting 95th percentile statistical significance thresholds. If no minimally reference sites are available, one could assume that a maximum score represented best attainable and set an impairment threshold as a percentage of the possible score (see e.g. Hughes et al. (1998), who used 75 percent of the possible score as an impairment threshold). A consequence of the percentile of reference approach to setting impairment thresholds is a tradeoff between Type I errors (concluding a site differs from reference when it truly does not) and Type II errors (failing to detect a that site differs from reference when it truly does). When setting a 5th percentile of reference as the impairment threshold, we are by definition making a 5 percent Type I error, since those reference sites below the 5th percentile are still from the reference site group. The smaller we set our Type I error rate, by selecting lower cutoff scores, the larger the type II error becomes. For rangeland streams, setting the impairment threshold at the 5th percentile of reference would include 70 percent of the impaired sites, or if the threshold were set at the 25th percentile, then 78 percent of the impaired sites would be detected (Table 4-6). Forest streams show a similar pattern, although the differences in Type II error are broader, ranging from 57 to 22 percent over the same range (Table 4-7). In both forest and rangeland streams, Type I and Type II errors were most balanced and their sums (total error term) were lowest at the 25th percentile of reference. Of course these comparisons assume that all reference sites are truly have few human disturbances, and all test sites have fish assemblages that were truly altered by humans, and all misclassifications are due to inaccuracies in the index. Instead, it could also be true that the index is correct and the sites were misclassified, or a combination of factors.

Table 4-6. Comparison of percentile distributions of fish index scores at forest reference and test streams.

Rank Among Reference Sites	Forest Reference Sites	Test Sites	% of Disturbed Sites with IBI <Reference Percentile	"Type II" Error Rate	"Type I" Error Rate	Combined Error Rate
n	230	47				
Minimum	0	0				
5th percentile	34	0	43%	57%	5%	62%
10th percentile	43	0	59%	41%	10%	51%
25th percentile	67	0	78%	22%	25%	47%
Median	81	41	88%	12%	50%	62%
75th percentile	91	63				
90th percentile	97	86				
Maximum	100	100				

Table 4-7. Comparison of percentile distributions of fish index scores at rangeland reference and test streams.

Percentiles of Site Scores	Rangeland Reference Scores	Test Sites	% of Disturbed Sites with IBI <Reference Percentile	"Type II" Error Rate	"Type I" Error Rate	Combined Error Rate
n	46	30				
Minimum	0	0				
5th percentile	39	0	40%	60%	5%	65%
10th percentile	49	0	58%	42%	10%	52%
25th percentile	62	18	74%	26%	25%	51%
Median	82	44	92%	9%	50%	59%
75th percentile	88	63				
90th percentile	94	77				
Maximum	100	92				

An alternate approach to setting impairment thresholds would be to set both an impairment threshold for the indexes, and a scoring scheme with other ecological indicators [macroinvertebrates and habitat], which are also scored on a 0 to 100 scale. For example, if any one index was below the 5th percentile of reference (or a percentile of all possible scores), the stream would be considered impaired, or both the fish and macroinvertebrate indices were below a percentile of reference (e.g. 10th, 25th, or xth), the stream would be considered impaired. This type of approach would both identify streams that had any one assemblage modified beyond the range of natural variability, and use macroinvertebrate, fish and habitat information together to assess stream quality. This approach could also be used with upper percentile scores to identify streams with exemplary biological integrity to protect from degradation (in water quality jargon, identify high quality streams in an antidegradation context).

Classification of Stream Types – Possible Refinements

Scale

The goal of this effort was to develop stream fish indexes with statewide or at least broad applicability. To meet this goal, I used a broad classification of streams as being either in the cool-desert, rangeland stream group or in the cold-mountain forest stream groups (Figure 4-20). Some additional scoring delineations by stream size and major river basin (to decide whether a species should be considered native in that basin), and stream size are included in the index scoring. However, streams in a large state such as Idaho could be further broken down by a host of additional factors, such as parent geology, stream geomorphology (e.g. Rosgen 1996), or additional climatic refinements. One has to consider a tradeoff between precision, and broad applicability. Indexes developed for smaller areas would likely have improved precision over a broadscale model.

According to Dunham et al. (ms.), "[t]he history of habitat modeling to predict abundance or occurrence of stream fishes has shown that models based on sampling a range of temporal and spatial scales often have low predictive ability, and that models with high predictive ability have low transferability across different times or places." Their observation would be germane to developing IBI models as well. Developing an IBI model with a smaller set of more homogenous streams would likely provide more precision than the current model, many idiosyncratic models would be needed to describe different streams or river basins. The current approach gains the generality required for efficient application over a wide variety of streams, at the likely cost of lower precision. If higher precision is necessary, for example, for impact assessment or restoration monitoring, a paired watershed or similarly geographically focused assessment scheme would likely be needed, such as a Before-After-Control-Impact (BACI) design (e.g. Smith et al. 1993). In that case, a more site-specific model could be useful in addition to, or instead of this model.

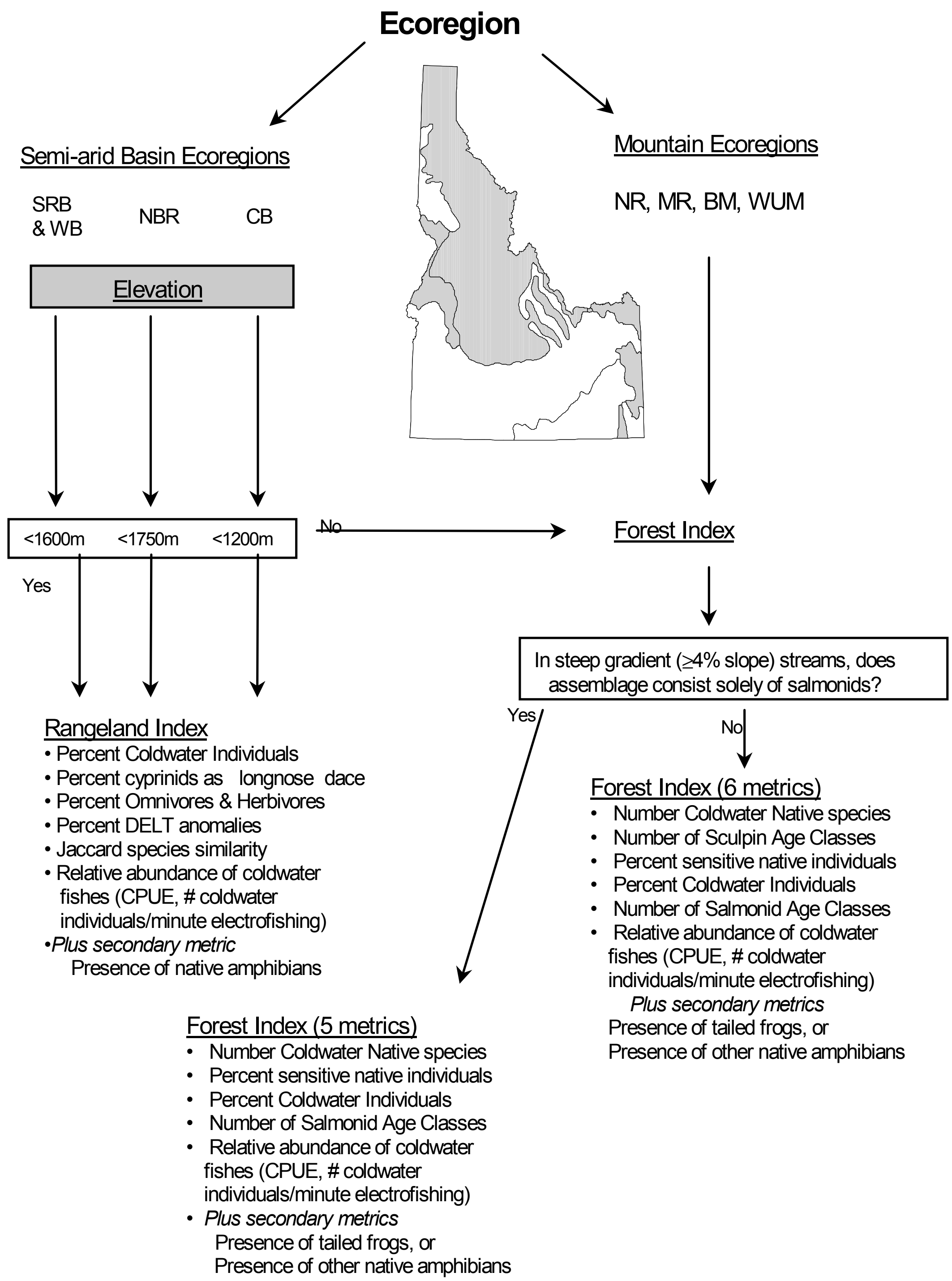

Figure 4-21. Flowchart of stream fish index structure.

Expected changes in ecoregion classifications

Omernik and his colleagues are in the process of revising and mapping more detailed ecoregions for Idaho. Among the changes are more detailed delineation of the current "Northern Basin and Range" ecoregion into sub-ecoregions showing "basins" and "ranges" (James Omernik, personal communication). Also, desert mountain ranges such as the Owhyee range in southwest Idaho would no longer just be considered part of the huge Snake River Basin/High Desert ecoregion which currently sprawls from the Idaho/Wyoming border to the eastslope of the Oregon Cascades (Omernik and Gallant 1986). In the present report, the delineation between high desert mountain streams and lower desert basin streams was added in a very coarse fashion through the addition of elevation as a factor determining whether a stream sample site would be expected to have a more montane biology or a lowland, basin biology (Figure 4-1, Figure 4-20). When the revised ecoregions are published, it would be interesting to overlay them with the site classifications shown in Figure 1, and consider whether this classification could be described (or should be revised) following the new ecoregions.

Numbers of fish in a sample: patterns and consequences

Two fundamental assumptions of a multimetric index based on attributes of a fish assemblage are 1) that there is a fish assemblage present and if so, 2) it was representatively sampled. At some point, one has to decide "how many fish in a sample are needed to suppose that the fish assemblage was adequately sampled?" Compared to fish samples from rivers or streams in more productive parts of the country, the numbers of fish per sample were relatively low with medians of 28 and 15 fish captured at rangeland and forest sites respectively. Oligotrophic, small mountain streams, in particular often had low numbers of fish captured. Fore et al. (1994) examined statistical properties of a warm water IBI used in streams in Ohio. They noted that Ohio samples with less than 400 fish tended to have more variable IBI scores. Clearly this rule of thumb for what constitutes a "small" number of fish in a sample would have to be revised in the forest-mountain streams of Idaho where the 95th percentile of numbers of fish caught in a sample was only 168.

Low numbers of fish in a sample could be the result of 1) inefficient sampling, 2) low densities from severely polluted conditions, or 3) low densities form marginal physical habitat conditions, such as steep gradients, low flows, or freeze-outs. After the fact, it is difficult or impossible to know which reason was responsible (although one might assume that meticulous or sloppy and incomplete datasheets reflects on sampling effort as well). Results from inefficient sampling should be rejected, yet a simple rule of rejecting samples with low counts of fish would also discard samples from which fish were rare, losing valuable assessment information. Sheldon (1987) writes that often stream fishes are rare, with absolute population numbers in the hundreds. Fragmentation and isolation of stream habitats due to flow removal, impassible culverts, or impoundments, for example, will increase the risk of local stream extinctions. Thus, flagging sites with unusually sparse fish numbers could be useful for identifying streams warranting follow-up investigation.

This still leaves the question of how many fish in a sample makes a sparse sample? I made some informal tests by entering hypothetical data for several common species associations with increasing sample sizes, increasing age classes, but maintaining relative species percentages. At less than 10 fish/sample, for different species combinations, a doubling of sample size resulted in score changes of up 18 points, between 10 and 20 fish, a doubling increased scores by about 8 to 11 points, when over 20 fish were in a sample, a doubling of sample size increased scores by about 3 to 7 points. When over 40 fish were in a sample, further increases the number of fish in a sample made little difference in the score. This exercise suggests that a sample with less than 10 fish should be flagged as sparse to caution the user.

CONCLUSIONS

Overall, the stream fish indexes for rangeland and mountain streams of Idaho show promise as tools to help interpret fish assemblages from a bioassessment perspective. The indexes should be complementary to the macroinvertebrate indexes (Jessup and Gerritsen 2000) and the habitat index (Fore and Bollman 2000). In order to be broadly applicable, the structure and calculations of the SFI are fairly complex, compared to most other IBIs (see articles in Simon 1999). However, this complexity was a factor complicating its development only, rather than its use. The decision rules and calculations are automated, and are transparent to the user (Figure 4-20). The output is a simple index value from 0 to 100, which can be dissected further, or left as a summary, depending on the interest level of the user.

Recommendations

The indexes in their present form could be useful; however, several short and longer-term recommendations could improve their utility.

Short term (1)

Investigate existing data for index variability and sensitivity. Because our site-numbering scheme, it was difficult to identify repeated sites. Assuming sites repeated at similar times of the year, without significant environmental perturbations should be similar, index values should be calculated and evaluated on these.

Long term (1)

Encourage interagency cooperation in sampling. Some USFS, BLM, and IDFG data sets from areas of interest where excluded because non-salmonids were not captured and recorded. Non-game fish are part of the stream community too. The rationalization that the salmonids are the most sensitive indicator species may not hold in all situations.

Long term (2)

Encourage consistent amphibian collection (or at least capture, documenting, and release). DEQ's extensive BURP program does not currently maintain a stream vertebrate database, only a fish database. Amphibian observations are relegated to field notes. This could easily be and should be rectified.

Long term (3)

Target reference streams to evaluate interannual variation.

Long term (4)

Investigate direct observation of fish through snorkeling. Well-meaning resource managers have restricted routine electrofishing surveys in their belief that they are avoiding population loss by not allowing electrofishing. Because of these perceptions, some salmonid population trends monitoring is done through snorkeling, with mixed results. Few examples have been reported of attempting to survey the entire assemblage, instead of just target species, or water column species. Pearsons et al (1992) and Torgerson (2000) reported some success using snorkeling to survey fish assemblages. This may be worth investigating, because of some attitudes toward electrofishing.

Long term (5)

Rate of anomalies may be a useful indication of environmental degradation, however both DEQ's current data and database management are weak in this area. USGS fish data tended to have higher rates of anomalies than did data form DEQ, IDFG or university collections. There are at least two explanations for this: (1) the USGS sampled more degraded waters than others, or (2) the USGS workers found more anomalies because they were specifically looking for anomalies instead of happening to notice them. Field crews should be trained on potential anomalies, and to take a few seconds to specifically examine each fish. Anomalies were more common on non-salmonids, such as suckers, which may receive less attention than the salmonids. Specific anomalies need to be tracked in the database, rather than just "anomalies" and details relegated to text fields, if at all. Blackspot disease and lip tumors do not have similar biological significance.

Longer-term (6)

Since this is an empirical effort, the metrics and index streams should be re-evaluated when more data are available, and revised accordingly, particularly for the rangeland streams.

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